PRODUCTION NOTE

ILLINOIS NATURAL HISTORY SURVEY

Large-scale Ecology of the Vermilion River Basin

Aquatic Biology Section
Technical Report

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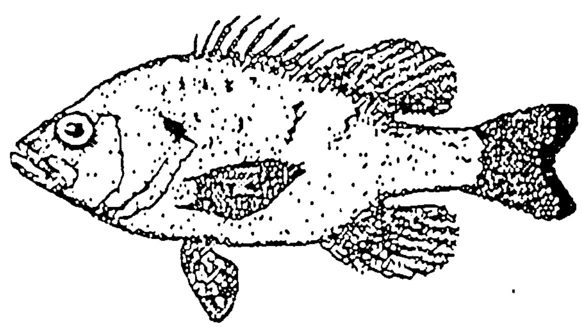
Final Report
ENR Project EH24



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PREFACE

Realization that the integrity of biological communities in streams and rivers is intimately tied to the surrounding landscape has resulted in the development of qualitative and descriptive lotic ecosystem models (*e.g.*, continuum hypothesis, resetting hypothesis). Unfortunately, descriptive models provide inadequate information for managing aquatic resources or for predicting possible impacts on stream systems associated with proposed watershed modifications. As water demands in Illinois grow, new pressures are placed on surface waters, and their protection and wise allocation becomes increasingly important. The State Water Plan explicitly recognizes two key issues related to management of our stream resources--erosion and water withdrawal. Addressing legitimate demands of society, while simultaneously maintaining sufficient quality and quantity needs of aquatic communities, necessitates a quantitative understanding of the complex linkages that exist between the stream system and the surrounding terrestrial environment.

The goal of this project was to study ways in which basin geomorphological, hydrological, and cultural characteristics influence large-scale patterns of biological productivity in stream ecosystems. Incorporation of basin characteristics into impact assessement procedures used to allocate Illinois water resources was emphasized.

This report is organized topically into chapters. Chapter 1 examines the history of land use in the Vermilion River drainage of east-central Illinois. Political and physiographic boundaries within the watershed are identified and examined from a historical perspective. Important socio-economic attributes of the watershed, including agriculture, livestock production, and mining are reviewed. Such major landscape modifications as stream channel modification, distribution of urban areas, and location of point source discharges are also examined. The above information was used throughout various stages of the study to facilitate data interpretation.

In Chapter 2 land-use patterns within the Salt Fork watershed were developed from aerial photographs and digitized onto the Prime computer. The buffering facility of the Arc/Info software was employed to generate land-use patterns at various distances from the stream channel. Nitrate-N and soluble reactive phosphorus (SRP) data, collected bi-weekly from 22 sampling stations, were related to watershed land-use patterns and empirical relationships were developed. These empirical relationships were then used in the development of an interactive model to predict the potential impact of proposed land-use alterations in the watershed on instream nitrate-N and SRP concentrations. In this chapter, we stress the importance of examining nutrient impacts from the perspective of instream concentration, and thus system integrity, rather than from the perspective of loading, which is more appropriate to impacts on downstream lentic systems.

In Chapter 3, land-use, geomorphological, and hydrological characteristics of the landscape are empirically examined in relation to nutrient availability and instream primary productivity of the stream system. We argue for, and demonstrate the use and application of, empirically constructed models of key biological variables combined with physical descriptors from hydraulic geometry as a basis for stream ecosystem theory and management. We argue that models which incorporate key, driving variables can provide a basis to predict and explain the diversity of structures, whereas generalized descriptive models cannot.

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Chapter 1

A HISTORY OF LAND USE IN THE VERMILION RIVER WATERSHED

THE STUDY AREA

Location and Size

The watersheds of the Middle Fork and Salt Fork, tributaries of the Vermilion River, in east-central Illinois were studied. The Salt Fork covers much of eastern Champaign County and west-central Vermilion County (Fig. 1-1). The Middle Fork watershed begins in central Ford County, extends into the northeast corner of Champaign County, and then into west-central Vermilion County (Fig. 1-1). The combined drainage area is 893 miles^2, of which 465 miles^2 are in the Salt Fork watershed and 428 miles^2 in the Middle Fork. The two watersheds are fairly compact, with a prominent arm extending northwest into Ford County.

Vegetation

Much of the watershed was naturally covered by prairie. For example, in Champaign County only 1 acre in 12 was forested, with the remaining area covered by various types of prairie (Hansen 1963). The forested areas were found primarily along water courses; Vermilion County had more forested lands than either Champaign or Ford counties.

The prairie contained marshes and ponds in numerous depressions and swales (Barker et al. 1965). Albert W. Herre, an 1860's naturalist, referred to Champaign County as "a prairie covered with water every spring" (Hansen 1963). A railroad worker in the 1860s described Champaign as "one vast pond whose green, scum-coated surface was crossed by the trail of the water-mocassin" (Hansen 1963). Both observations portray an area with little natural drainage. Drainage in Ford County was also poor, with some areas covered by 6 ft of water (Ames 1970). Much of this county was swamp with the flies and mosquitoes reportedly so bad that farm work could only be done by moonlight (Ames 1970).

Virtually all the prairies and marshes have been converted to agricultural lands. Forested areas have also been reduced by cutting, grazing, disease, and fire. In Champaign County, the forests had been reduced from an estimated 50,000 to 7,000 acres in 1963.

HISTORY OF THE WATERSHED

Dramatic changes have occurred in the study area since presettlement times, including draining wetlands for agriculture, altering stream courses, and urbanizing areas. Each alteration has impacted water quality. To understand the effect of these anthropogenic changes, we must examine the history of the watershed. The Middle Fork and Salt Fork watersheds encompass a large portion of Champaign, Vermilion, and Ford counties. Because of the large area involved and due to the availability of census data, information is provided for each county.

Champaign County

Champaign County's name was derived from the word 'champaign,' which means level, open country (Hansen 1963). Runnel Fielder first settled the area; he built a cabin and began farming 4 miles northeast of Urbana in 1822. Settlement was slow, with roughly a dozen families moving into the area from Kentucky by 1830. At that time, the land was part of Vermilion County. The population in the western part of the county increased to several hundred by 1833, at which time Champaign County was created by the State Legislature. Most of the settlement occurred on the Salt Fork where timber was available for construction and firewood.

The U.S. Census of 1840 listed a population of 2,981 for Champaign County (Table 1-1). In general, the county experienced steady growth to a population of 168,392 in 1980. There were, however, periods of strong growth. When the railroad was built, the population increased from 2,649 in 1850 to 14,629 in 1860 and to 32,737 in 1880. The pace of growth slowed until 1940-1950, when the census reported an increase from 70,578 to 106,100. Growth again slowed between 1970-1980.

Champaign and Urbana, the two major cities in Champaign County, experienced steady growth, with one period of rapid growth. Between 1940 and 1950, Champaign increased from 23,302 to 39,563 and Urbana from 14,064 to 22,834. In the 1860 census, Champaign had a population of 1,727; by 1980, it was 58,133. Urbana's population was 210 in 1850, increasing to 35,978 in 1980.

Vermilion County

The availability of farmland and timber were important to the settlement of Champaign County, but salt was the attraction in Vermilion County. The first reference to this area comes from a 1706 French record mentioning the "Salines of the Vermilion" (Williams 1930). In November 1819, Seymour Treat and his family settled approximately 6 miles west of what is now Danville, in an area referred to as the "Salt Works" (Williams 1930). The salt was first class in quality and purity, attracting settlers from Indiana and from the Illinois River. Salt production was the start of industrial development in the Danville area. In 1824, Major John W. Vance, from Ohio, leased the "Vermilion Salines." The Salt Works, located at the mouth of Stoney Creek, used 80 140-gallon kettles to boil water brought up from 50- ft deep wells to produce salt. The salt industry was short-lived, however. Discovery of the Sciota salt fields in the 1830s ended the "Vermilion Salines." The wells were abandoned in 1840.

The short-lived salt industry did attract settlers. The Salt Works were originally part of Edgar County. On 18 January 1826, Vermilion County was created. At the time of organization, it included parts of what are now Ford, Iroquois, Livingston, Grundy, and Will counties. In 1859, the county took its present shape and size. In 1821, the population was 200 (Williams 1930). By the 1830 census, the population of the county was 5,836 (Table 1-2). Vermilion County experienced more erratic population changes than did Champaign County. The population increased steadily to 89,339 in 1930, declined to 86,791 in 1940, grew slowly until 1970-1980 when the population fell from 97,047 to 95,222.

Danville is the major city in Vermilion County. The city was planned as a river city but, because of the Vermilion River was not deep enough, it never materialized. Despite the failure of the salt works and the inability to develop as a river city, Danville grew to

4,751 in 1870 due to the discovery of coal reserves. By 1872, the coal mining operations of A. C. Daniels was the city's chief industry and most of the residents were employed in the coal mines (Williams 1930). Danville's population grew steadily to a peak of 42,570 in 1970, but decreased by 1980 to 38,985.

Ford County

Ford County was the last county organized in Illinois. On 17 February 1859, it became Illinois' 102nd county. Paxton, formerly called Prospect City and Prairie City, was named the county seat. Ford County was settled much later than either Champaign or Vermilion counties, due to the extremely wet conditions throughout much of the county. John Cooder, Joseph Coontz, and David Patton settled Ford County in 1849 (Gardner 1908). The 1860 census lists a population for Ford County of 1,979 (Table 1-3). Growth was rapid until 1880 when the population reached 15,099. Then population changes became erratic. The population peaked in 1900 at 18,359, declined to 15,007 in 1940, stabilized, increased to 16,382 in 1970, and decreased to 15,265 in 1980.

There are no large cities in Ford County. Paxton, the county seat, and Gibson City are approximately the same size, with a 1980 population of 4,258 and 3,498, respectively. Each has grown steadily from an 1870 population of 1,456 in Paxton and 1880 population of 1,260 in Gibson City.

LAND USE IN THE WATERSHED

A logical consequence of settlement and increased populations was a dramatic change in land use. Forests, prairies, and wetlands were replaced by farm fields, pastures, and cities. The relationship between land use and stream ecosystems is well documented. As native vegetation was converted to crops, sedimentation of surface waters became a problem; pollution resulted from mining activities, industrial effluents, and urban activities.

Land use in the watershed can be divided into two categories--agricultural and urban. The percentage of watershed under cultivation, livestock production, artificial drainage, and the use of fertilizers and pesticides are agricultural activities that affect water quality. Disposal of urban garbage and sewage effluents and mining operations also affect water quality.

Agricultural Land Use

The availability of farmland attracted many early settlers to Champaign, Vermilion, and Ford counties. The conversion of native vegetation to cropland was the first major land-use change. Changes in agricultural practices also have significantly affected water quality. Through an examination of census data, several trends in agricultural practices in all three counties are evident:

1. Acreage in farmland increased rapidly after settlement, peaking around 1900. There was a slight decline and a stabilization since that time.

2. The number of individual farms peaked at approximately the same time as did the number of acres in farmland. There has been a sharp and steady decline in the number of farms since 1900.

3. Since settlement, crop diversity has been significantly reduced. Livestock production also has declined.

4. There has been steady increase in the use of fertilizers and pesticides. Current use is highly dependent on weather conditions and crops.

5. Most drainage activities took place from 1860 to 1900. Most recent work has involved maintenance of the channels.

Farmland. According to the 1850 census, Champaign County had 22,873 acres of improved farmland, with a total of 58,173 acres in farms (Table 1-4). By 1860, improved acreage increased to 169,610. A second major increase in improved farmland occurred in the next decade, when the number of acres rose to 419,368. The acres in farmland peaked at 622,613 in 1880. The acres in farmland in Champaign County remained near 600,000 from 1880 to the present. In 1982, the county had 600,159 acres in farmland. The total acreage in the county is 638,784, or 94% of the county is farmland. The number of farms peaked in Champaign County in 1880, followed by a steady decline. The 1880 census reported 5,022 farms, declining to 4,316 in 1900 and 3,108 in 1950. Only 1,871 farms were reported in 1982.

Vermilion County, settled earlier than Champaign or Ford counties, reported 147,382 acres in farmland in 1850 (Table 1-4); only 11,759 acres were listed as improved. Improved acreage was 247,867 by 1860, 360,251 by 1870, and peaked at 575,182 acres in 1900. Since 1900 the number of acres in farmland has declined to approximately 520,000 acres and has remained fairly constant. In 1982, Vermilion County reported 499,920 acres in farmland. Total acreage for the county is 576,006, or 86.8% of the county is farmland. The number of farms peaked early and then declined steadily. In 1900, Vermilion County had 4,138 farms, declining to 3,202 in 1950, 1,814 in 1969, and 1,390 in 1982.

The 1860 census lists 32,591 acres of farmland in Ford County, with 16,155 as improved acreage. By 1870, over 200,000 acres were reported as farmland, 141,228 of that improved. Improved acreage increased to 308,455 in 1900, declined to 217,776 acres in 1940. This number increased to 309,607 acres in 1969. By 1982 the number of acres in farmland was listed as 289,155. With a total land area of 310,822, 93% of the county is currently farmland. In 1880, there were 2,138 individual farms in Ford County, 1,901 in 1900, 1,377 in 1950, and 804 in 1982.

Crop Production. Crop diversity was greatest prior to 1900 in each county. Wheat, rye, "Indian corn," oats, barley, buckwheat, hay, Irish and sweet potatoes, and even tobacco were grown. Tobacco was not a significant crop, however, peaking in Vermilion County in 1860 at 17,127 pounds. By 1900, barley, buckwheat, and tobacco were no longer reported, or were grown in very small quantities. Rye also decreased in importance. Corn was the dominant crop. Soybeans were not reported in any county until 1920. After 1950, soybean production became a significant factor. By 1982, wheat, corn, oats, soybeans, and hay were the only crops reported in large quantities. An emphasis on primarily two crops, corn and soybeans, since the 1960's has significantly impacted water quality, including sedimentation, use of chemicals, and loss of riparian vegetation.

Champaign County has followed the pattern of decreased diversity. Wheat production was 1 million bushels in 1920, 1959, and 1969. In 1982, 372,763 bushels were reported. Rye peaked at 67,742 in 1880 declining to 980 bushels in 1900. Rye continued to be grown in small quantities until 1978 when it was eliminated from the census.

Corn, or "Indian corn" as it was referred to in the censuses until 1900, has always been the dominant crop of Champaign County. In 1850, 441,060 bushels of corn were grown in Champaign County; production increased steadily with a significant increase between 1959 (16,121,300 bushels) and 1969 (26,414,561 bushels). In 1982, corn production climbed to 40,746,979 bushels.

Oats have had erratic production levels in Champaign County, with 38,850 bushels reported in 1850. Production levels peaked at 7,339,520 in 1900, declining steadily to 1,367,523 in 1940. There was an increase to almost 4 million bushels in 1950, followed by a crash in production. Only 106,874 bushels of oats were reported in 1982 for Champaign County. Barley and buckwheat were never grown in large quantities. Barley peaked at 58,679 bushels in 1930, dropped to 1,906 bushels in 1950, and has not been reported since 1969. Buckwheat production peaked in 1860 at 9,478 bushels and was not reported after 1930. Hay production in Champaign County averaged 40,000 tons for 1870-1959, declining to 11,240 tons in 1982. This decline is related to decreased livestock production in the Champaign County.

In Champaign County, Irish and sweet potato production both peaked in 1870 at 266,918 and 2,263 bushels, respectively. Irish potato production declined steadily to 1,167 bushels in 1959 and was not reported after 1978. In 1950 only 217 bushels of sweet potatoes were reported and have not been reported since.

Soybeans (4,109 bushels) were first reported in the 1920 census for Champaign County. Production increased dramatically to 3,192,854 bushels in 1950 and to 10,648,075 in 1982, becoming one of the two dominant crops in the county.

Vermilion County. Crop diversity was highest in Vermilion County prior to 1900 until only hay, wheat, corn, oats, and soybeans were reported in 1982. Wheat production (46,301 bushels) was first reported in 1850. Production increased to 294,364 bushels in 1870 and remained fairly constant until 1930, when it increased to almost 700,000 bushels. The 1-million bushel level was surpassed in 1950-1969 but production in Vermilion County declined to 284,727 bushels in 1982. Rye production in Vermilion County has been highly variable. Only 427 bushels were reported in 1850, increasing to 52,476 in 1870. Production declined to 3,520 bushels in 1900 but rose to 73,714 in 1920. By 1969 rye was no longer listed in the census for Vermilion County.

Corn has always played an important role in Vermilion County agriculture. Production in 1850 was triple that of Champaign County at 1,475,195 bushels. Between 1870 and 1880, production increased to 6,385,086 bushels, remaining fairly constant until 1978 when production was over 18 million bushels. In 1982 production increased to almost 30 million bushels of corn in Vermilion County.

Oat production in Vermilion County has lagged behind that of Champaign County since 1850. In 1860, production was 88,181 bushels, peaked at 4,541,230 bushels in 1900, and declined to 58,630 in 1978. Barley production from 1850 to 1900 averaged 4,255 bushels, with a low of only 60 bushels. Peak production was 43,117 bushels in 1930. No barley is reported after 1959 in Vermilion County. Buckwheat reached its highest production level in 1860 at 13,670 bushels, declining to 45 bushels by 1930. No record is given after 1930. Hay production in 1870 was 52,553 tons, remaining fairly constant until it dropped to 17,942 tons in 1950. Production increased to almost 37,000 bushels by 1959 but decreased to 12,063 tons in 1982.

Production of Irish potatoes in Vermilion County peaked at 175,558 bushels in 1870 but declined steadily thereafter. In 1959, 734 bushels were reported but no production was reported in later years. Sweet potatoes were never a major crop although 55,533 bushels were reported in 1880. Typical annual production during from 1850 to 1940 was 1,000 bushels; no record is available after 1950.

As in Champaign County, soybeans were first reported in 1920 when only 411 bushels were produced. Production rapidly increased to 3,186,154 bushels in 1950 and to over 8 million bushels in 1982.

Ford County experienced similar patterns in crop production as Champaign and Vermilion counties, although slightly later. Production levels are substantially less than either Vermilion or Champaign counties, but the land area is half that of Champaign County. Wheat production has been highly variable--from 350 bushels in 1900 to 215,472 bushels in 1920. By 1982 production declined to 78,409 bushels. Production of rye was 18,110 bushels in 1880, declining to 300 bushels in 1900. No production figures were listed after 1940.

Corn has been the dominant crop in Ford county. In 1870, 565,671 bushels were reported, increasing to 4.5 million bushels in 1880. In 1978, the county produced 26.5 million bushels, but just over 18 million bushels in 1982.

Oats (5,889 bushels) were first reported in Ford County in 1860, increasing to 2,260,865 by 1890. Production remained over 3 million bushels until 1940, when it declined to under 2 million bushels. Oats remained a stronger crop in Ford County than in the other two counties, with 1982 production at 90,392 bushels. Barley and buckwheat were insignificant crops in Ford County. Barley ranged from 360 to 5,690 bushels from 1860 to 1930, when it increased to over 44,000 bushels. Barley declined to 1,540 bushels in 1940 and was not reported in later censuses. Buckwheat peaked at 702 bushels but was not reported after 1900. Hay production remained fairly constant from 1870 (23,445 tons) to 1940 (26,000 tons). Production declined steadily to 5,923 tons in 1982.

In 1860, 6,692 bushels of Irish potatoes were reported in Ford County, increasing to 107,244 bushels in 1890. Production declined to 16,907 bushels in 1940 and to 8 bushels in 1969. The highest production of sweet potatoes was 671 bushels in 1900 and declined to 80 bushels in 1950. After 1950, sweet potato production was dropped from the census.

Soybeans (697,398 bushels) were not reported until the 1950 census in Ford County. Production increased to over 7 million bushels by 1978, before declining to just over 5 million bushels in 1982.

Livestock Production. The three counties experienced similar patterns in livestock production. In 1980, each county reported production for horses, asses and mules, milch cows, ox, other cattle, swine, and sheep. By 1982, this had changed to horses, sheep, swine, and cattle (beef and dairy). The number of species raised decreased by 1982, and the numbers were greatly reduced as well, reflecting a greater dependence on soybeans and corn.

Horses were found in large numbers in all three counties until 1940, when numbers dropped sharply. The peak was 30,926 in Champaign County in 1910, 27,279 in Vermilion County in 1890, and 16,675 in Ford County in 1890. By 1982, Champaign

County reported 963, Vermilion County 570, and Ford County 185. Asses and mules were never found in large numbers in any of the three counties. Almost 3,000 were reported in Champaign County in 1900 but declined to 562 in 1940. In Vermilion County, there was 2,205 head in 1850 but 82 in 1950, when they were dropped from the census. Ford County reported a high of 1,528 in 1880, decreasing to 252 in 1940.

Cattle production records are complicated by changes in categories from one census to another. Dairy cattle were more common in Champaign County prior to 1920 than they are today. A peak of 21,854 head was reported in 1920 but only 681 in 1982. Dairy cattle also peaked in Vermilion County in 1920, at 17,619 head, declining to 413 in 1982. Dairy cattle were not as common in Ford County, with 9,762 head in 1920 and 681 in 1982. Beef cattle were listed separately in the census after 1920, so total cattle figures are the best indications of overall cattle production. In 1920 ,there were 35,890 head in Champaign County, declining to 20,060 in 1969 and 11,058 in 1982. Vermilion County raised 42,822 head in 1950 but 16,302 in 1982. Ford County had 26,760 cattle in 1959 but 15,087 in 1969 and 7,354 in 1982.

Sheep production in Champaign County ranged from 8,000 head to 18,000 head from 1860 to 1959. Production declined to 6,239 in 1969 and 2,069 in 1982. Vermilion County sheep production in 1860 and 1870 was approximately 70,000 head and declined to under 20,000 head by 1920. Production fluctuated between 8,000-15,000 until 1978, when production dropped to 2,238 and to 2,069 in 1982. Sheep production peaked in Ford County in 1969 at 8,085 head. Production then fell to 2,845 in 1969 and 1,254 in 1982.

In 1860, 36,384 swine were reported in Champaign County, increasing to 119,092 in 1890 and declining to just over 43,000 in 1910. Levels ranged from 30,000 to 60,000 until 1982, when production declined to 28,721. In Vermilion County, a peak of 108,558 swine was reported in 1890 but decreased to 18,431 in 1900. Swine production ranged from 37,038 to 74,008 from 1910 to 1978. In 1982, 45,921 swine were reported in Vermilion County. Swine production in Ford County followed a similar pattern. In 1890, 52,926 head were reported and in 1940 only 18,576. The 1982 census reported 34,551 swine in the county.

Chemicals and Fertilizers. Information regarding the use of pesticides and fertilizers is dispersed and unreliable. In some cases, state or national production levels are available but not for specific counties. In 1959, the U.S. Census added information on the use of commercial fertilizers, herbicides, insecticides, and lime. With records available for so few years, it is impossible to estimate change, but it is possible to estimate the degree to which these chemicals are used in each county. Their impact on water quality depends on numerous factors unrelated to the chemicals themselves, such as weather, ground preparation, slope, and amount used per acre. Use of fertilizers has increased in each of the three counties in terms of both acres and tons used (Table 1-5). For example, Ford County used fertilizers on 70,616 acres in 1959 and 162,554 in 1982. Herbicides are widely used and increasing in use. Champaign County used herbicides on 267,179 acres in 1969 and on 469,721 acres in 1982 (Table 1-6). Insecticide usage is far below that of herbicides. Use of insecticides appears to have declined in Champaign and Ford counties from 1969 to 1982 but have increased slightly in Vermilion County. Champaign and Ford counties increased their use of lime from 1959 to 1969 (Table 1-7), but usage decreased sharply in 1982. Tonnage decreased from 38,881 in 1969 to 18,983 in 1982 in Ford County.

Artificial Drainage

Surface-water resources have changed in all three counties since settlement. Marshes, ponds, and streams were typical during pre-settlement times. Now there are streams, many of them artificially created or dredged, and man-made ponds and lakes; nearly all marshes having been drained. These major drainage improvements have been crucial to the development of agriculture in the watershed.

The technology was not available for adequate drainage of Ford and Champaign counties until 1860-1870, when land drainage experts arrived from Europe (Barker et al. 1965). Use of ceramic tile by these German immigrants allowed thousands of acres of wet prairie and marsh to be converted into fertile cropland.

The first attempt at drainage was digging ditches using a mole-ditching machine. This machine resembled a large plow, with a wedge of iron 3-4 ft in length, attached to a sharp blade. The machine was pulled by two yoke of oxen and could ditch 0.5 mile/day. The ditch provided temporary relief from wet conditions.

Michael Sullivant purchased 80,000 acres in 1855, much of it in the Salt and Middle Fork watersheds. The land was extremely wet, with some pieces under 6 ft of water. Sullivant used a ditching plow 18 ft long, with a plowshare 11 ft x 2 ft x 10 in. It required 68 oxen and 8 men to operate it. This machine increased the speed of digging to 3-3.5 miles/day.

Drainage Districts. Most of the stream channels in the three counties were artificially created after 1880. Opening these drainage ditches caused problems for downstream landowners, which resulted in disputes between neighbors. This situation posed a legal problem because the Illinois constitution did not provide for enactment of drainage laws. In 1870 and 1878, the constitution was revised to include drainage provisions. The legislature subsequently passed the Farm Drainage Act and the Levee Act in 1879. These acts provided for the organization of drainage districts, the appointment of commissioners, surveys for drains, assessment of taxes based on benefits to be derived, and maintenance. Township commissioners were empowered to approve drainage districts formed under the Farm Drainage Act, whereas the courts supervised those formed under the Levee Act. As a rule, the land near major rivers was organized under the Levee Act and the upland districts under the Farm Drainage Act. Approximately 65% of the drainage districts were organized under the latter act, for which information is more difficult to obtain (Hay and Stall 1974).

Since passage of these acts, over 100 amendments have been added and dozens of court rulings have been made regarding the provisions of each act, leading to confusion and complexity. On 1 January 1956, the Drainage Code became effective (Hay and Stall 1974), replacing the Levee and Farm Drainage Acts. It gave authority to the Circuit Court, required annual reports to be filed by each district, and gave authority to the county treasurer to manage the funds. This has simplified the work of the districts to some extent.

Drainage Events. Efforts to drain Ford, Champaign, and Vermilion counties can be divided into three periods (Hay 1974): (1) Earliest settlement to 1880 was characterized by individual and mutual ditching but little organized effort involving large areas. Ditching methods were slow and primitive, with the shallow channels often inadequate. Disputes with downstream landowners were common. (2) The Farm Drainage Act and the Levee Act were passed in 1879, providing for the formation of drainage districts. From 1880 to 1956, the drainage districts constructed many of the channels that exist today. (3) The new drainage district law or Drainage code became effective in 1956, replacing the Farm Drainage and Levee Acts and placing authority for their supervision with the courts. Maintenance has been the primary activity from 1956 to the present.

Drainage District Activities. Considering the size of the Vermilion River watershed, a greater proportion of agricultural drainage has occurred in this watershed than in any other in Illinois. Most drainage districts were formed immediately following passage of the drainage laws in 1879. There was an intensive period of ditch digging, channel straightening, and dredging. Although records of this early period are available, the actual location of the work undertaken is not well defined. This problem continues today. Drainage districts are required to file an annual report, but these reports generally contain total expenses and receipts for each year, not specific activities. Besides drainage districts responsible for the main channels, numerous subdistricts were formed that constructed and maintained branch ditches; much of this ditching was done on or near farm fields. In drainage district records, only the activities of the main districts were considered. The activities along the major channels would likely impact water quality more than those actions in farm fields a greater distance from the channel.

We had to determine if the district was active or inactive. An active status is gained by filing legal documents appointing commissioners to the district. Because drainage districts are a function of the courts, an inactive district may become active by designating the drainage commissioners to the circuit clerk's office. Many drainage districts revert to an inactive status until maintenance of the main channel is necessary. Commissioners are then elected and fees assessed to do the work.

A second component of a district's status is whether annual maintenance fees are assessed. Whatever the annual costs for legal fees, engineering fees, herbicides, and other administrative tasks, these are divided among the landowners of the district. If additional maintenance is necessary, such as dredging or channel straightening, the district files a request for an additional assessment with the court. These fees are also divided equally among the landowners.

The active or inactive status indicated in Tables 1-8 through 1-10 means the district elected commissioners for 1986 and that an annual maintenance fee was collected. This assessment is often a nominal amount, used to pay attorney fees and insurance. In Champaign County, many districts have large budgets or regularly file for additional assessments. Those districts marked by an asterisk (Tables 1-8 to 1-10) indicate major channel improvements were done within the past 6 years.

In Champaign County, practically all of the Vermilion River watershed is divided into organized drainage districts. Champaign County has more large drainage districts than any other county in Illinois. These districts were responsible for most the drainage channel construction in the county. Thirty districts are active, and 20 have been involved in major channel activities in the past 6 years. District 2 (Beaver Lake), covering 35,000 acres in the western reaches of the watershed, filed for an $38,741 in 1981 and had a 1985 budget of $47,158, which covered only 40% of the year's expenses. The district has been involved in brush clearing, tile repair and cleaning, silt removal, and other improvements. The Raup District (Number 24) petitioned for $37,000 in additional funds in 1982 to open 4 miles of ditch, including removal of shrubs, dredging, and correcting the slope of the channel banks. No major channel work had occurred in that district for over 30 years. These districts are two of the more active ones, but the districts highlighted in Table 1-8 are involved in similar activities. It is impossible to determine where each district has dredged or altered the stream channel.

Vermilion County has less acreage in the Salt and Middle Fork watersheds included in drainage districts, in part, because of topography, particularly in the Middle Fork area. Few districts were formed in the Middle Fork watershed. Sixteen districts are active (Table 1-9) and five involved in major drainage activities. District 25 (Jordon Special) petitioned for additional funds in 1980 to open 6 miles of ditch and to repair or clean 8.5 miles of tile drain. No major work has been undertaken since that time. Stoney Creek (District 39) has been involved in recent activities, although no major maintenance or repair has occurred since 1950. In 1980, the district petitioned for an additional levy of $195,000 to clean, dredge, and clear the main channel and its branches and to repair tiles. The remaining three active districts have undertaken similar projects. Channels are dredged, cleared, and cleaned of brush. Silt is removed from the drain tiles and repaired where necessary.

Ford County has seven drainage districts located in the Middle Fork watershed, and one is the fourth largest in Illinois. Big Four (District 1) has jurisdiction over much of the western portion of the watershed. Five districts are active and three have been involved in major drainage improvements in the past 6 years, affecting 54,000 acres. In the past 6 years, Big Four District has excavated ditches, cleared brush, and aerially sprayed riparian vegetation. In 1979, the repairs to the channel were estimated to require a $600,000 levy. Big Four has numerous subdistricts, which are also active. In 1980, District 4 (No.1 of Lyman Township) cleaned the main channel, in 1981 draglined some areas, and in 1983 cleared the main channel again. Several districts, although not involved in major improvements, budgeted funds for herbicides to keep the banks clear of vegetation. No record is available of the type or quantity of herbicide used.

Urban Land Use

Urban activities have increased dramatically in the past 50 years. Land uses important to water quality are the percentage of the watershed paved, sewage treatment facilities, waste disposal, existing industrial effluents, and mining activity. Data are not readily available on either the surface runoff due to paved areas or on the production of industrial effluents. Paving decreases the absorption capabilities, thereby increasing the flow of sediment and urban pollutants into the stream.

Sewage Treatment Plants. At one time all sewage effluents were released with no treatment into streams. By 1938, only Champaign and Urbana treated their sewage before disposal in streams (Regional Planning Commission 1938). Today, many towns in the watershed treat their sewage effluents, although rural areas and small villages rely on septic tanks.

In Champaign County, there are six sewage treatment plants located in the watershed--two in Urbana-Champaign and Rantoul and one in Gifford and St. Joseph. Data on these facilities include design, annual and low flow, permit dates, and rates of biological oxygen demand (BOD), total suspended solids (TSS), and ammonia (NH_3), including any violations for these three standards.

For 1983-1985, three facilities in Champaign County violated at least one standard. Gifford had violations in 1983 and one in 1985; two were excess BOD and TSS violations. One Rantoul plant violated the BOD standard once in 1984 and twice in 1985. The Urbana-Champaign NE plant also violated the BOD standard four times in 1983 but operated within standards in 1984 and 1985. The Urbana-Champaign plant experienced significant differences in designed versus lowest flows recorded. The design flow was 17.30 mgd, but the 3-month low flow was 11.220 mgd. Four plants in Champaign County provide tertiary treatment. The Urbana-Champaign facilities use high-rate filters, Gifford an intermittent sand filter, and Rantoul-West a polishing pond. Three types of main treatment are used in these plants. Four use activated sludge, Rantoul East trickling filters, and Gifford a lagoon.

Vermilion County has six sewage treatment plants in the watershed. Only one has experienced violations during 1983-1985. The Fithian plant had exceeded the TSS standard twice in 1983. Of the six plants identified, information is available for five. The Potomac plant has no data on flow, violations, or type of treatment. Four plants provide tertiary treatment. Catlin and Danville use high-rate filters, Oakwood a submerged sand filter, and Fithian an intermittent sand filter. The plant not using tertiary treatment uses a trickling filter as the main treatment. Catlin and Danville use activated sludge and Oakwood and Fithian use a lagoon. The Danville plant suffered dramatic low flow periods. The design flow was 16.0 mgd, with an annual flow of 8.910. The 3-month low, however, was 6.666 mgd.

In Ford County, there is one treatment plant in the watershed. The Paxton plant received its permit in 1982 and has had only one violation between 1983-1985; it exceeded the TSS standard in 1984. The Paxton facility uses activated sludge as the main treatment and high-rate filters for tertiary treatment. The design flow of the system was 0.6 mgd and had a 3-month low flow of 0.4 mgd.

Primary Sewage Treatment. There are three main processes used by these sewage treatment plants:

The activated sludge process has been successfully used to treat domestic sewage. The process involves biological floc contacting liquid waste in an aerated system. Organic matter is removed by sludge microrganisms that oxidize the organics in the presence of air. The process mixes primary settled sewage with activated sludge and aerates the mixture for 4-8 hours in long, narrow basins. The mixture of activated sludge and sewage flows into a settling tank; clarified liquid is discharged at the top and sludge solids are removed in the underflow. A portion of the sludge is returned to be mixed with inflowing sewage and the remainder is removed from the system.

Stabilization lagoons have been used successfully, especially in rural areas. They use principals of natural purification. Organic matter is decomposed by a combination of aerobic, facultative, and anaerobic bacteria. Bacteria and algae in stabilization lagoons form a symbiotic relationship that quickens the treatment. Bacteria aerobically stabilize the organic matter, releasing carbon dioxide. The algae assimilate carbon dioxide, releasing oxygen for the bacteria. The principal problems of lagoons are odors, excessive algal growth, high algal suspended solids in the final effluent, and a dependence on climatic conditions.

Trickling filters are widely used in biological treatment of sewage and industrial wastes. These filters are shallow, circular tanks filled with crushed stone, wooden slats, or plastic media. Settled liquid waste is applied over the surface of the filter by a rotating distributor and is collected and discharged at the bottom. The filters do not remove organic matter. Rather organic matter is adsorped by slimes covering the filter media. Removal of BOD and suspended solids range from 80-90%.

Tertiary Treatment. Filtration is the most commonly used tertiary process. It is generally less expensive than some other methods and can be used to "polish" secondary effluents to meet standards. A rapid sand filter strains flocculents and sediment. Particles are trapped between grains of the filter media, primarily at the filter's surface.

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Table 1-1. Population of Champaign County and the cities of Urbana and Champaign, 1830-1980. NA = not available.

Year	Champaign County	Champaign	Urbana
1830	—	—	—
1840	2,981	NA	NA
1850	2,649	NA	210
1860	14,629	1,727	NA
1870	32,737	4,625	3,325
1880	40,863	5,103	2,942
1890	42,159	5,839	3,511
1900	47,622	9,098	5,728
1910	51,829	12,421	8,245
1920	56,959	15,873	10,244
1930	64,273	20,348	13,080
1940	70,578	23,302	14,064
1950	106,100	39,563	22,834
1960	132,436	49,583	27,294
1970	163,281	56,532	32,800
1980	168,392	58,133	35,978

Table 1-2. Population of Vermilion County and the city of Danville, 1830-1980. NA = not available.

Year	Vermilion County	Danville
1830	5,836	NA
1840	9,303	NA
1850	11,492	736
1860	19,800	1,632
1870	30,388	4,751
1880	38,588	7,733
1890	49,905	11,491
1900	65,635	16,354
1910	77,996	27,871
1920	86,162	33,776
1930	89,339	36,765
1940	86,791	36,919
1950	87,079	37,864
1960	96,176	41,856
1970	97,047	42,570
1980	95,222	38,985

Table 1-3. Population of Ford County and the towns of Gibson City and Paxton, 1830-1980. NA = not available.

Year	Ford County	Gibson City	Paxton
1830	---	---	---
1840	---	---	---
1850	---	---	---
1860	1,979	NA	NA
1870	9,103	NA	1,456
1880	15,099	1,260	1,725
1890	17,035	1,803	2,187
1900	18,359	2,054	3,030
1910	17,096	2,086	2,912
1920	16,466	2,234	3,033
1930	15,489	2,163	NA
1940	15,007	2,401	3,106
1950	15,901	3,029	3,795
1960	16,606	3,454	4,370
1970	16,382	3,454	4,373
1980	15,265	3,498	4,258

Table 1-4. Acres of farmland in Champaign, Vermilion, and Ford counties.

Year	Champaign	Vermilion	Ford
1850	58,173	147,382	---
1860	146,806	383,595	32,591
1870	494,650	424,451	208,200
1880	622,613	547,322	276,786
1890	598,335	532,586	290,408
1900	627,785	575,182	308,455
1910	608,428	534,385	304,019
1920	604,827	519,338	295,972
1930	608,375	529,335	298,706
1940	597,066	514,211	217,776
1950	604,900	521,577	296,334
1959	600,519	501,713	299,226
1969	605,282	514,922	309,607
1978	592,571	506,542	305,631
1982	600,159	499,920	289,155

Table 1-5. Commercial fertilizer used in Champaign, Vermilion, and Ford counties, 1959-1982. NA = not available.

	Champaign		Vermilion		Ford	
Year	Tons	Acres	Tons	Acres	Tons	Acres
1959	32,617	243,115	29,155	225,749	10,200	70,616
1969	52,232	240,119	40,255	193,175	22,200	118,106
1982	NA	327,528	NA	266,302	NA	162,554

Table 1-6. Herbicides and insecticides used in Champaign, Vermilion, and Ford counties, 1959-1982.

	Champaign		Vermilion		Ford	
Year	Herbicide	Insecticide	Herbicide	Insecticide	Herbicide	Insecticide
1959	NA	NA	NA	NA	NA	NA
1969	267,179	99,301	212,663	73,248	121,595	46,372
1982	469,721	76,614	367,401	75,173	221,079	24,587

Table 1-7. Agricultural lime used in Champaign, Vermilion, and Ford counties, 1959-1982.

	Champaign		Vermilion		Ford	
Year	Tons	Acres	Tons	Acres	Tons	Acres
1959	27,564	13,182	41,902	19,649	26,755	12,563
1969	35,964	15,215	44,713	18,912	38,881	15,001
1982	36,621	18,071	38,132	17,255	18,983	9,098

Table 1-8. Champaign County drainage districts.

Number	District	Activity	Area (acres)	Date organized
1	Bailey Branch	Inactive	1,600	1892
*2	Beaver Lake	Active	35,276	1880
3	Big Tile Ditch, County	Inactive	1,200	1901
4	Buck Creek Mutual	Inactive	2,315	1928
5	Conkey Branch Special	Active	3,260	1908
6	Dillsburg Special	Active	3,088	1917
*7	No. 10, Town of Ogden	Active	789	1900
8	No. 11, Town of Ogden	Inactive	647	1901
9	No. 1, Town of Sidney	Inactive	2,274	1898
10	No. 2, Town of Sidney	Inactive	2,185	1908
*11	No. 1, Town of S. Homer	Active	4,125	1893
*12	No. 3, Town of St. Joseph	Active	6,293	1880
*13	No. 4, Town of St. Joseph	Active	4,619	1880
14	No. 5, Town of St. Joseph	Inactive	410	1881
15	No. 8, St. Joseph Township	Inactive	865	1910
*16	No. 1, Stanton Township	Active	2,070	1908
17	Ehman-Schmidt Mutual	Inactive	1,050	1928
*18	Flatville Special	Active	7,202	1909
19	Hickory Grove	Inactive	NA	1920
20	Kerr & Compromise	Active	1,835	1913
21	Killbury Mutual	Inactive	60	1924
22	Mutual, Harwood Township	Inactive	1,420	1880
23	Mutual, Harwood Township	Inactive	490	1882
*24	Raup	Active	3,078	1926
*25	Salt Fork	Active	7,151	1904
*26	Saline Branch	Active	16,182	1906
27	Schindler	Inactive	566	1906
28	Schneider	Inactive	849	1925
*29	Silver Creek	Active	5,088	1909
*30	Spoon River	Active	21,302	1903
*31	South Fork	Active	3,137	1903
*32	St. Joseph No. 6	Active	720	1881
*33	Stanton Special	Active	3,973	1908
*34	Triple Fork	Active	4,175	1929
35	Union, Stanton & Ogden	Inactive	2,150	1900
*36	Union, No. 1 Ogden & Oakwood	Active	13,000	1880
37	Union, No. 1 Philo & Urbana	Active	2,284	1907
*38	Union, No. 2 Somer & Stanton	Active	9,117	1901
39	Union, No. 1 S. Homer & Sidell	Inactive	2,110	1882
40	Union, No. 2 S. Homer & Sidney	Inactive	4,380	1894
41	Union, S. Homer & Sidney	Active	1,120	1896
*42	Union, No. 2 St. Joseph & Ogden	Active	3,931	1880
43	Union, No. 7 St. Joseph & Ogden	Inactive	840	1909
*44	Upper Salt Fork	Active	9,668	1925
45	Urbana & Champaign Sanitary	Inactive	5,503	1934
46	West Branch	Active	1880	1906

Table 1-8 (concluded).

Number	District	Activity	Area (acres)	Date organized
47	Willow Branch	Active	1,005	1902
48	Wrisk	Active	1,736	1904
49	Youman's Branch Mutual	Inactive	1,679	1929
50	Harwood & Kerr	Active	3,958	1925
51	Ludlow Special	Inactive	2,540	1948
52	Somer Township No. 1	Active	2,300	1950

*Drainage improvements have been made since 1979.

Table 1-9. Vermilion County drainage districts. NA = not available.

Number	District	Activity	Area (acres)	Date organized
3	Bean Creek	Active	5,226	1895
8	Butler Branch	Active	2,800	1918
9	Center Creek	Active	1,263	1921
10	No. 2, Vance Township	NA	330	NA
11	No. 1, Blout Township	NA	NA	NA
12	No. 1, Town of Ross	NA	1,420	NA
15	Eight Mile	Active	5,452	1911
16	Feather Creek Union 1	Active	1,630	1906
*17	Feather Creek No. 2	Active	1,605	1925
18	Grape Creek	Active	3,216	1915
20	Hammel Mutual	Inactive	437	1925
21	Henning	Inactive	894	1912
*23	Jamesburg Special	Active	4,207	1912
24	Johnson	Active	5,038	1925
*25	Jordan Special	Active	8,623	1906
26	No. 1, Oakwood Township	Active	1,520	1908
27	No. 4, Oakwood Township	NA	NA	NA
28	No. 6, Oakwood Township	NA	NA	NA
29	No. 7, Oakwood Township	NA	980	1980
30	No. 8, Oakwood Township	NA	930	NA
31	No. 9, Oakwood Township	NA	150	NA
32	No. 12, Oakwood Township	NA	438	1919
34	Pleasant View	Active	2,018	1918
35	Rose Township Mutual	NA	1,215	1925
37	Sinking Hole	Active	2,015	1913
38	Special, Vermilion & Champaign Co.	NA	422	1895
*39	Stoney Creek	Active	11,076	1906
43	Union No. 1, Oakwood & Vance	NA	740	NA
44	Union No. 1, Vance & Catlin	Active	2,164	1910
45	Union No. 1, Vance & Sidell	Active	8,545	1880
46	Westville	Inactive	363	1920

*Drainage improvements have been made since 1979.

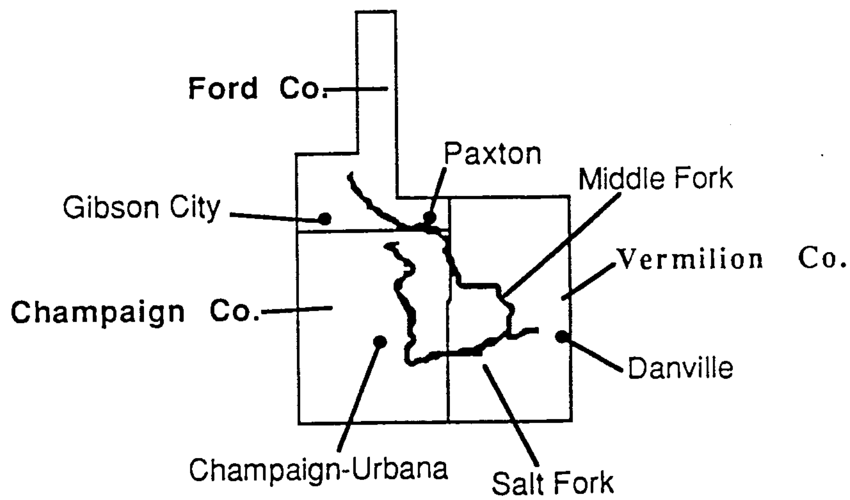

Fig. 1-1. The Middle Fork and Salt Fork, tributaries of the Vermilion River, in east-central Illinois.

Chapter 2

EMPIRICAL RELATIONSHIPS BETWEEN LAND USE/COVER AND STREAM WATER QUALITY IN AN AGRICULTURAL WATERSHED

INTRODUCTION

During the past two decades, extensive economic and physical resources have been devoted to restoring and protecting surficial waters (lakes and streams) in the United States. This expenditure of resources has coincided with increasing understanding of the complex processes involved in the maintenance of healthy, functioning ecosystems and in the mechanisms of degradation. Initially, water quality protection and restoration programs, as mandated in the Clean Water Act (P.L. 92-500), concentrated on control of point-source inputs. Despite the intensified efforts of these control programs and the implementation of advanced control technologies, recent studies (*e.g.*, Roseboom et al. 1982) indicate that established water quality standards will not be achieved in many regions because of non-point sources of pollution, which were not addressed in earlier programs.

The significance of diffuse sources to water quality degradation (Loehr 1974, Uttormark et al. 1974, Browne 1981) necessitated alternate methodologies to assess the impact of physiographic and land-use characteristics on water quality in streams. As in any new area of research, a plethora of approaches evolved, including local examinations of sediment and nutrient loadings in small watersheds (Roseboom et al. 1982, Bonini et al. 1983); use of USGS topographic maps and aerial photographs to calculate land-use patterns in a watershed (Omernik 1976); and relating data on fertilizer use and application, agricultural sales, forests, and artificial drainage to prevailing water quality in a single river basin (Craig and Kuenzler 1983). In the latter case, the amount of agricultural land was determined by cutting and weighing maps.

The localized experimental approach, as employed by Roseboom and others, enhanced our understanding of water quality degradation (*e.g.*, surface runoff and pollutant transport) and provided valuable empirical measurements of sediment loading associated with a given land use. However, such studies are often beyond financial and time constraints of local or regional management and planning agencies. Less expensive, more rapid, and more diversified techniques applicable to a wide range of geographical areas were needed, which led to several studies on the effects of large-scale patterns of land use on stream water quality.

A study by Omernik (1976), although contributing to a general understanding of the effects of land use on water quality in the eastern United States, did not provide sufficient information to determine the amount of forest required to significantly reduce inputs of watershed nutrients in stream channels. Such information is directly applicable in planning restoration of disturbed lands that juxtapose streams of marginal quality. Further, significant physiographic differences between the eastern United States (*e.g.*, slope, soil type, and geologic history) and other geographical regions limit the utility of such large-scale studies in managing and restoring local streams. The methods employed by Craig and Kuenzler (1983), although relatively inexpensive, relied on potentially incomplete government records, changing data collection and reporting methodologies through time, the assumption of an even distribution of farmland within each geopolitical boundary, and the inherent inaccuracy of cutting and weighing of maps.

This cursory review of techniques indicates the limited methodologies available to address land-use effects on stream water quality. Further, the above discussion reflects the problem of scale in extrapolating environmental data to many planning and management issues, and more specifically, the "open ended" nature of stream ecosystems and their intimate relationship to the watershed and associated anthropogenic uses. Minimally, empirical relationships between background land use and cover patterns and water quality in the watershed are needed to develop successful management strategies.

This chapter presents a methodology to facilitate planning decisions and the formulation of development policies for enhancing stream water quality at the watershed level. Additionally, we examine the empirical relationships between land-use practices and stream nutrient concentrations in a Midwestern watershed using a computer-based Geographic Information System (GIS). Successful control programs and policies will only be achieved if environmental planners and managers address water quality degradation from a watershed scale. Such a perspective, by definition, includes point and diffuse sources.

LOADING VS CONCENTRATION

It is important to differentiate between the effects of land use on subsequent instream concentrations and the effects on loading. Calculations of loading are more appropriate in determining impacts on downstream areas, particularly receiving waters such as lakes or reservoirs (Schlosser and Karr 1981). Loading has been typically used by engineers and those concerned with reservoir management. From a toxicological and biological perspective, instream concentrations are considered more appropriate due to the obvious relationship between prevailing abiotic conditions and stream community structure. Schlosser and Karr (1981) suggest that, resources permitting, both loading and concentration should be determined. In resource limited situations, the relative emphasis should vary depending on the water resource problem.

Due to the close ties with engineering, the classical approach in environmental planning has been to apply deductive models (*e.g.*, Streeter-Phelps) and view effects of land use on water quality from the perspective of loading. Loading need not be related in any way to the health or "integrity" of the stream and its associated communities. Therefore, if the management objective is to maintain the quality of the stream, rather than downstream reservoirs, concentration measures should provide a more meaningful indication of integrity. Within the context of this report, water quality will be viewed from the perspective of instream concentration.

STUDY AREA

The study was conducted in the watershed of the Salt Fork, a major tributary of the Vermilion River, in east-central Illinois (Fig. 2-1). This watershed encompasses an area of roughly 500 miles^2, mostly in row-crop production (primarily corn and soybeans). Two major urban areas occur in the watershed, the twin cities of Champaign-Urbana (approximately 100,000 population) located in the western portion of the watershed and the city of Rantoul (approximately 35,000 population) located in the northern most section of the watershed (Fig. 2-1). The Chanute Air Force training facility is located on the southern fringe of Rantoul. Three other smaller municipalities (St. Joseph, Oakwood, and Sidney), each with populations less than 5,000, are distributed along the middle and lower reaches of the main branch of the Salt Fork. Municipal effluents (mostly secondary facilities) from these areas are discharged into the river. One major gravel mining operation is located in the southeastern most portion of the watershed (Fig. 2-1).

METHODS

Water Chemistry

Water chemistry data were collected from 22 stations (Fig. 2-2) on a bi-weekly basis from December 1983 through December 1984. Samples were collected during storm and non-storm events, providing representative concentrations during the study. Summer (July-September) was characterized by low discharge and high temperatures; spring (March-June) by high flows, precipitation events, and moderate temperatures; fall (October-December) by generally low flows with moderate temperatures; and winter by low flows with periods of complete ice cover.

Water quality parameters measured were nitrate-N, ammonia-N, nitrite-N, turbidity, minimum and maximum biweekly temperatures, specific conductance, pH, and soluble reactive phosphorus (SRP). The analyses discussed here will be restricted to nitrate-N and SRP. SRP is essentially equivalent to dissolved phosphorus and represents material that is immediately available for use by autotrophic organisms. Total phosphorus was collected once to determine the relative proportion of SRP to total phosphorus in the stream.

Land-Use/Cover Patterns

Land-use/cover patterns were determined by interpreting false-colored infrared National High Altitude aerial photographs (1:20,000 scale; film exposure 1983) obtained from the EROS Data Center, U.S. Department of Interior, Sioux Falls, SD. Land-use/cover patterns were interpreted onto mylar sheets and digitized on a PRIME 750 computer. ARC/INFO (Environmental Systems Research Institute), a geographic information system that manipulates data in digital form, was used to analyze the data.

Because of the nature of the research being addressed, it was determined that the optimal classification system would include both use and cover categories. "Use" refers to man's activities that are directly related to the land (Clawson and Stewart 1965) and "cover" describes vegetation and artificial construction covering the land surface (Burley 1969). The classification system employed was a modified version of the LUDA system (Anderson et al. 1976), which is a resource-oriented rather than social- or human-oriented classification system (Guttenberg 1965). To make the results as widely applicable as possible, the modified classification system had six general categories: urban and built-up land, agricultural land, forest land, water (lakes, reservoir, and streams), wetlands, and barren lands (*e.g.*, mining areas).

Land-use/cover patterns were digitized as polygons (enclosed areas), stream networks as line coverages, and stations as point coverages. All digitized coverages were converted to a standardized geographic reference scale (Lambert feet) and merged with adjacent coverages of the same type (*i.e.*, polygon, line, or point) to form a single coverage representing the land-use/cover patterns, stream network, and water quality stations in the Salt Fork watershed.

Watershed boundaries for each water quality station were determined using 7.5-min (1:24,000) U.S.G.S. topographic maps and digitized. These boundary files were employed to cut out the land-use/cover patterns within each station's watershed using the ARC/INFO "clip" function, which resulted in a total of 22 watershed areas containing both land-use/cover patterns and stream networks.

Buffer Regional Analysis

Land-use/cover patterns within four distances (100, 200, 400, and 1,000 ft) of the stream channel were determined using the ARC/INFO "buffer" facility (Fig. 2-3). Essentially, the buffer facility constructs polygons whose outer boundaries parallel the stream network (line file). The linear distance or width of the outer boundary from the line (*i.e.*, stream reach) is set by the operator. The total area (in acres) of each land-use/cover category (Table 2-1) within each water quality station's watershed was determined using the computer software. The total area of each land-use/cover category within each individual buffer area was used to create land-use buffer regions of <100, 100-200, 200-400, 400-1,000, and >1,000 ft.

The mean concentrations of nitrate-N and SRP for each season were calculated for each station and related to watershed area and the land-use/cover patterns in each buffer region using multiple regression analysis (Ryan et al. 1982). Land-use/cover areas for each water quality station were converted to proportions of the total station area within each buffer region to facilitate standardization and help control differences in total areas encompassed by each station. Ratios of urban to agriculture (UR/AG) and forested to agriculture (FR/AG) and total square miles of area within each station watershed were used as the independent variables. Prior testing indicated no significant correlation between the independent variables, and there was no significant spatial auto-correlation in the data set. Thus, the components of the regression model for each water quality parameter and buffer region were mean concentration of the water quality parameter at each station (dependent variable) and the independent variables: the station's total watershed area (reflecting general position within the watershed) and the UR/AG and FR/AG ratios. Data on buffer region land use and on water quality for each chemical sampling station were fit to this model and used to generate a regression relationship for each water quality parameter and buffer region (Table 2-2).

RESULTS

Land Use/Cover

The Salt Fork watershed (Fig. 2-1) included 281,516.7 acres of land, approximately 90% of which was used for agriculture (Table 2-1). Urban areas were roughly 5% of the total watershed area while forested areas were approximately 2.5%. The remaining land (<2.5%) was distributed among lakes, barren lands, and wetlands. The breaks in the headwater hydrology (Fig. 2-2) are a result of drainage modifications, including installation of sub-surface tiles.

Examination of the land-use/cover map (Fig. 2-1) demonstrates that agriculture areas were evenly distributed throughout the watershed. The urban and forested areas were patchy, with the largest urban areas located in the upper reaches of the watershed and the forested areas in the lower reaches adjacent to the main river channel. In the lower reaches of the Salt Fork watershed, more than 50% of the total forested acres were within 200 ft of the stream channel. Thus, while forests are limited within the watershed, the majority can be considered riparian in nature. The absence of riparian forests in headwater areas departs from the classical concept of watershed structure (Cummins 1979, Vannote et al. 1980).

Classically, stream watersheds are considered to consist of narrow headwater channels with well developed forested canopies that grade into wider stream channels downstream with little or no forested canopy. Riparian vegetation minimizes bank erosion and limits inputs of nutrients from the surrounding land. This downstream transition is also thought to produce corresponding changes in water quality, such as increased temperatures, increased nutrient loadings, increased bank erosion, and decreased energy inputs from outside the system. Such a concept of watershed structure has dominated the management and restoration literature (Gore 1985), particularly as related to the establishment of water quality goals and mitigation techniques (Herricks and Osborne 1985). The paucity of information on structural and functional differences between Midwestern watersheds, such as the Salt Fork, and more "typical" watersheds has hindered past regional management and restoration efforts.

Despite these differences, some pertinent similarities exist, particularly in the location of urban areas. Historically, urban areas were located along stream channels because of the importance of the river network for trade and travel. Tributaries of the Mississippi River still serve this basic function. Furthermore, streams provide convenient disposal for urban wastes. All major urban centers and most of the smaller communities in the Salt Fork watershed are adjacent to a stream channel. While urban areas are <5% of the total land use/cover in this watershed, their proximity to stream channels and the density of the human populations could adversely impact water quality.

Water Chemistry

The results of the instream concentrations of SRP and nitrate-N with respect to watershed location are graphically presented as 3-dimensional watershed profiles (Fig. 2-4). The y-axis is months, the x-axis is station drainage area in square miles, and the z-axis (grayscale) is the concentration in mg/L of the parameter. Use of these figures provide a rapid assessment of spatial and temporal variability of measured instream parameters in the watershed.

The high nitrate and SRP concentrations reflect the eutrophic nature of the Salt Fork watershed (Fig. 2-4). SRP concentrations generally increased from headwaters to downstream reaches, suggesting there is nutrient loading in the main channel (Fig. 2-4).

A mean maximum reactive phosphorus concentration of 6.13 mg/L was recorded in fall immediately downstream of the city of Urbana. The lowest mean concentration (0.07 mg/L) was recorded in spring in a tributary in the lower portion of the watershed. During the study, the mean maximum SRP concentrations were always recorded downstream of Urbana, despite tertiary treatment of municipal sewage. Stations located downstream of other urban areas also had substantially higher concentrations of SRP than did more rural stations (Fig. 2-4).

The highest SRP concentrations generally occurred during low-flow periods (*i.e.*, late summer-early winter) while the lowest concentrations occurred during higher discharge periods, suggesting that there is a negative relationship with stream discharge (and thus, precipitation). The lower concentrations during high flows were likely a result of dilution. The higher concentrations from fall through winter are apparently a result of lower discharge (less dilution) and a lower rate of instream primary productivity (Wiley and Osborne, unpublished data), which would decrease the amount of SRP taken up by autotrophs.

With such a spatial chemical data set, it is not possible to determine if higher SRP concentrations downstream of urban centers is attributable to point-source effluents or to non-point source urban runoff (Blake-Coleman 1984). However, both non-point and point sources are associated with urban land-use activities. SRP concentrations were always higher downstream of urban centers, even during major precipitation and runoff periods (spring). Unlike previous reports of the adverse effects of agricultural activity on stream nutrient loadings (Craig and Kuenzler 1983), these results suggest that effects of agricultural practices on instream SRP concentrations are minimal compared with urban influences in the Salt Fork watershed.

Nitrate-N concentrations were generally highest from late winter through spring (Fig. 2-4). During this same period, nitrate concentrations were high throughout the entire watershed. From summer through early winter, nitrate concentrations were lower in the headwater reaches, with periodic pulses of higher concentrations downstream in the vicinity of urban areas (Fig. 2-4). From late winter through spring, nitrate concentrations were at their maximum in the agriculturally dominated headwaters, indicating that nitrate-N concentrations are seasonally impacted by agricultural activity.

The higher concentrations in late winter and spring coincides with nitrogen (ammonia) fertilization of agricultural fields in the watershed. Nitrogen is transported to the stream via surface runoff and subsurface flow associated with spring rains and snowmelt. Prior to entering the stream, most ammonia is converted to nitrates (Wiley, Osborne, and Larimore, unpublished data). As primary productivity is lower during this period, nitrate concentrations remain fairly high despite the greater discharge. With increasing instream primary production and crop development (and therefore uptake of nutrients applied to the field), nitrate concentrations decrease except downstream of urban areas. Thus, a spatial analysis of the chemical data suggests that urbanization is primarily responsible for increased SRP concentrations in the Salt Fork watershed and also for increased nitrate-N concentrations during summer and fall. Agricultural practices appear to largely affect the concentration of nitrate-N from late winter through spring.

Empirical Relationships

Three independent variables (station watershed area, UR/AG, and FR/AG) explained a highly significant amount of the variation associated with SRP concentrations in the Salt Fork watershed during all periods examined (Table 2-2). Similar relationships were found for all seasons and buffer regions for nitrate-N, except in the winter when statistically insignificant relationships were found in all buffer regions >100 ft from the stream channel (Table 2-2).

During spring, summer, and fall, the UR/AG variable accounted for most of the explained variance associated with nitrate-N concentrations (Table 2-2). In winter, the FR/AG variable accounted for most of the explained variance, although the regressions were not significant. The FR/AG variable also contributed to a substantial proportion of the explained variance in spring. In summer and fall, the UR/AG variable accounted for the majority of the explained variance in nitrate-N concentration (Table 2-2). The positive UR/AG coefficient during summer and fall further suggests that urbanization, rather than agriculture, contributed to the explanation of variance, consistent with spatial analysis results of chemical data. The lower relative value of UR/AG coefficients in winter and spring (Table 2-2) and the increased contribution of the FR/AG coefficient to the total explained sum of squares further substantiates the reduced importance of urbanization impacts on stream nitrate-N concentrations during these periods. This decrease is probably due to extensive application of nitrogen fertilizers in the watershed.

The general lack of importance of the FR/AG variable in the regression analyses for both SRP and nitrate-N was surprising, given the proximity of forested lands to the stream channel in the lower portion of the watershed (Fig. 2-1). The benefits of riparian vegetation in reducing nutrient inputs and bank erosion have been conclusively demonstrated (Anderson and Ohmart 1985). The limited importance of riparian forests in the present study, however, appears to be attributed to (1) their restriction to the lower portion of the watershed and (2) the overriding influence of urbanization on SRP concentrations. The riparian forests in the lower portions of the watershed limit nutrient inputs from the immediate surrounding areas but has no effect on mitigating inputs upstream. Further, riparian vegetation is of little value in controlling nutrient loadings if the principal inputs are from point sources, such as municipal treatment plants. These results reflect the open-ended nature of streams and the need for a watershed perspective in management and planning, as opposed to a localized perspective in attempts to mitigate stream water quality.

Undoubtedly, the majority of non-point source nutrient "loading" occurs during peak precipitation events and, within agriculturally dominated watersheds, most of the load originates with agriculture runoff (Omernik 1976, Craig and Kuenzler 1980). These facts have led to an emphasis on control of agricultural input of nutrients to stream channels. Despite the dominance of agricultural land use in the Salt Fork watershed, our results demonstrate that urbanization, rather than agriculture, has the greatest impact on stream SRP concentrations. Such findings are not surprising during low flows when urban point sources likely dominate SRP channel inputs. During spring high flows, however, urban areas still accounted for a significant proportion of the variance associated with SRP concentrations (Table 2-2). Nitrate-N appears to be associated with agricultural practices during the late-winter and spring, but then becomes more closely related to urbanization during the summer and autumn low-flow periods. Therefore, planners and watershed managers must be aware not only of the dynamic nature of stream systems but also of the temporal variability and relationships between components of the watershed.

PLANNING AND MANAGEMENT APPLICATIONS

These analyses, like others (Craig and Kuenzler 1983), provide insight into the effects of land-use practices on stream water quality. More importantly, however, the multiple regression models and buffer analyses can be used to address issues of concern to environmental planners and others interested in watershed management. One simple, but useful, application of these models is to examine the relative sensitivity of water quality variables to alterations in land use made at varying distances from the stream channel. Discriminating between potential effects of proposed land-use activities at alternate locations within a watershed using empirically derived models could be a powerful planning tool.

For example, we examined the potential impact of a proposed conversion of 100 acres of agricultural land to urban use on mean summer SRP and nitrate-N concentrations in the vicinity of Rantoul, Illinois (Fig. 2-5). The effects of such a conversion were determined for a stream site located approximately 8 miles downstream of the proposed urban construction. The drainage area was 48 miles^2. The existing land-use configuration and the proposed land-use/cover configuration (urbanizing an additional 100 acres) at five possible buffer distances from the stream channel were substituted into the empirically derived models (Table 2-3). The concentrations resulting from the proposed land use were then expressed as a percent change relative to the concentrations obtained with the existing land use and plotted against buffer distance from the stream channel (Fig. 2-5). The result is a diagram of the sensitivity of SRP and nitrate-N to land-use alterations at increasing distances from the river channel.

Mean summer SRP concentrations would be more sensitive than nitrates to increasing urbanization in the Salt Fork watershed (Fig. 2-5). Further, the impact of the proposed action diminishes with increasing distance from the river channel, implying that increases in buffer width would have relatively large mitigating effects on water quality. Additional development at a distance >1,000 ft from the stream would contribute little to SRP and nitrate-N concentrations in the downstream channel.

These empirical relationships between land use and water quality are specific to the Salt Fork watershed, although the same kind of analyses and methodologies could be applied to any water quality variable of interest in other watersheds. Many factors, (including terrain, type of vegetation, soils, and climate) contribute to geographic variation in watershed relationships (Barton et al. 1985). The validity of extrapolating the above empirical land-use/water quality relationships to similar watersheds in a region, or even to an adjacent watershed, is being studied. The incorporation of these methodologies into local planning or management could significantly improve siting decisions and regional water quality management programs.

APPLICATION AS A WATER QUALITY MANAGEMENT SYSTEM

The generation of regression equations can be thought of as a condensation of two large data bases--water quality and land use/cover. Such equations provide a convenient format for incorporating quantitative analysis into environmental management systems or similar planning software. A complete description of land-use/water quality relationships for each season in the Salt Fork watershed can be stored in a 160 x 4 element array of regression coefficients. High-speed, low-overhead access to average water quality characteristics at a specified site could be generated from simple row offset calculations based upon a query of the user's interests and needs. Such a software system could be extremely valuable to planners working on land-use alterations and/or impact mitigation. Interactive analyses of the effects of various land-use scenarios would allow evaluation of a number of siting alternatives, mitigation strategies, and long-term site development planning in a fast and relatively user-friendly gaming environment.

As an example of how empirical descriptions might be used in an interactive computer system environment, we developed a short demonstration program. The output from such a session is presented in Table 2-4. This software system implements a small subset of the regression relationships, providing an interactive analysis of the effects of land-use alterations on nitrate and SRP concentrations during summer low-flow conditions. An operator can enter the current land-use configuration, make proposed changes in that configuration, and then have the system estimate the likely changes in water quality at any designated location in the Salt Fork watershed.

Many organizations and agencies would not be able to generate as extensive of an array and frequency of water chemistry samples as used here to assess the potential impact of a proposed land-use alteration on stream water quality. In such situations, data from U.S.E.P.A., U.S.G.S., or state monitoring agencies could be obtained and used in the analyses. Similarly, land-use characteristics could be obtained from U.S.G.S. topographic maps or maps/photos maintained by local or regional planning commissions. Use of such information would greatly reduce the need for expensive data collection but would not alleviate the need for adequate computer software. While ARC/INFO is maintained on a mainframe, micro-computer packages are now available (Henco, Inc.) which could facilitate similar types of analyses by smaller agencies. Acquisition and/or construction of land-use and environmental data bases by local and regional agencies would provide more rapid and accurate assessment and enhance environmental management.

DISCUSSION AND SUMMARY

Interspatial and temporal analyses of chemistry data in conjunction with minimal knowledge of the location of major urban and agriculture areas provides the capacity to relate effects of gross human land-use activities to stream water quality. Other important management and policy issues, such as:

(1) what specific land use and land cover types are associated with changes in the concentration of individual water quality parameters;

(2) what is the amount of each land use or cover type that makes a significant difference in the concentration of a specific water quality parameter;

(3) how close to the stream channel does each land-use/cover type affect water quality; and,

(4) what is the quantitative relationship between land use/cover and a stream water quality parameter

are less evident with only a chemical data base. To address these management issues, a detailed analysis of land-use/cover types in the watershed and their relation to water quality conditions must be made.

The results of this investigation indicate that land use and cover significantly affect stream water quality and that viable management tools can be developed by integrating water quality data with buffer type analyses. The extension and application of such techniques would provide local and regional planning and management personnel with the capacity to reevaluate siting alternatives and provide information on location and sizes of mitigation procedures required to meet water quality goals.

For several years, federal and state environmental agencies have emphasized techniques and policies for controlling non-point source pollution. In the Midwest, most non-point source research has been directed towards controlling agricultural effects, with little emphasis on the effects of urbanization. Our results suggest that urbanization is the most important land-use activity controlling seasonal SRP concentrations, and thus water quality degradation, in the Salt Fork watershed. Significant impacts of urbanization were detected on SRP concentrations throughout the year and on nitrate-N concentrations during approximately half of the year. Therefore, the present research emphasis on agricultural impacts in Midwestern streams may be misplaced if the objective is to reduce instream nutrient concentrations. If the objective is to reduce loading in downstream reservoirs, then agricultural non-point source controls should be implemented as numerous other studies have documented. Thus, goals must be implicitly clear at the outset. A first step is determining if instream concentrations are to be reduced to meet water quality standards or if loading rates are to be reduced to limit eutrophication rates in downstream reservoirs.

Finally, this study provided valuable information on the general structure of land use/cover in a Midwestern watershed. The results of our analyses indicate substantial differences between the "classical" concept of watershed structure and that which occurs within many Midwestern watersheds. Thus, application of restoration and management techniques designed for use in more "classically structured" watersheds may not yield intended results in Midwestern watersheds due to simple, yet significant, differences in land use and cover structure.

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Table 2-1. The total acres of land within each level-I land use/cover category for each sampling station in the Salt Fork River basin.

Station	Agriculture	Urban	Forest	Lake	Barren
13	28,240	4,085.9	14.8	2.3	82.6
15	145,061	23.6	0.0	0.0	0.0
18	14,883	153.7	24.5	12.4	0.0
19	2,296	0.0	0.0	0.0	0.0
21	47,997	4,104.1	46.3	2.3	82.6
22	25,820	234.0	92.8	12.4	0.0
11	26,241	7,179.8	786.3	79.2	0.0
12	30,634	7,265.7	921.2	88.0	0.0
84	124,144	12,304.5	1,366.6	115.0	124.3
86	141,837	12,983.9	1,765.5	118.5	134.9
89	174,294	13,230.5	3,107.7	255.6	452.1
90	11,382	118.9	120.7	0.0	43.5
91	194,526	13,319.7	4,111.6	268.7	237.0
34	196,970	13,444.9	5,097.6	276.9	259.8
92	3,934	0.0	0.0	0.0	1,233.6
96	9,836	201.2	22.0	0.0	1,545.9
95	9,980	201.2	39.5	0.0	1,554.2
28	7,343	0.0	31.8	0.0	14.7
29	6,028	0.0	40.9	0.0	0.0
30	20,223	0.0	94.1	2.4	14.7
33	42,810	765.0	752.9	94.4	26.9

Table 2-2. Results of multiple regression analyses on seasonal mean soluble reative phosphorus and nitrate-N concentrations for five land-use/cover buffer regions. The independent variables are watershed area (SQMI), the ratio in urban acres to agriculture acres (UR/AG), and the ratio of forest acres to agriculture acres (FR/AG). * significant at $P<0.05$; ** significant at $P<0.01$; *** significant at $P<0.001$; NS is not significant; - denotes negative relationship.

Buffer	SQMI	UR/AG	FR/AG	Total SS	R^2 (%)
			Soluble Reactive Phosphorus		
Spring					
<100'	3.9	18.6	0.00	23.9	94.1 (***)
100-200'	3.9	18.6	0.03	23.9	94.4 (***)
200-400'	3.9	18.6	0.04	23.9	94.3 (***)
400-1000'	3.9	18.7	0.01	23.9	94.8 (***)
>1000'	3.9	14.1	2.48	23.9	85.8 (***)
Summer					
<100'	43.9	51.5	0.15	99.4	96.1 (***)
100-200'	43.9	52.5	0.10	99.4	97.1 (***)
200-400'	43.9	52.8	0.13	99.4	97.4 (***)
400-1000'	43.9	51.0	0.39	99.4	95.9 (***)
>1000'	43.9	49.8	1.68	99.4	94.9 (***)
Fall					
<100'	38.5	44.9	-1.43	89.5	94.9 (***)
100-200'	38.5	46.5	-1.27	89.5	96.5 (***)
200-400'	38.5	46.7	-2.14	89.5	97.7 (***)
400-1000'	38.5	44.3	-2.48	89.5	95.3 (***)
>1000'	38.5	45.0	0.39	89.5	93.8 (***)
Winter					
<100'	5.3	30.7	0.08	38.2	94.5 (***)
100-200'	5.3	31.0	0.04	38.2	95.3 (***)
200-400'	5.3	30.9	0.03	38.2	94.9 (***)
400-1000'	5.3	30.2	0.06	38.2	93.2 (***)
>1000'	5.3	25.7	3.13	38.2	89.5 (***)
		Nitrate-N			
Spring					
<100'	-3.5	6.5	3.84	25.1	55.1 (**)
100-200'	-3.5	6.0	3.65	25.1	52.6 (**)
200-400'	-3.5	6.6	5.32	25.1	61.3 (***)
400-1000'	-3.5	6.5	5.19	25.1	60.6 (**)
>1000'	-3.5	3.6	5.17	25.1	49.0 (**)

Table 2-2 (continued).

Buffer	SQMI	UR/AG	FR/AG	Total SS	R^2 (%)
Summer					
<100'	9.5	26.8	0.23	40.5	90.2 (***)
100-200'	9.5	27.3	0.01	40.5	91.0 (***)
200-400'	9.5	27.6	0.17	40.5	92.4 (***)
400-1000'	9.5	26.9	0.38	40.5	90.9 (***)
>1000'	9.5	25.2	0.57	40.5	87.2 (***)
Fall					
<100'	7.1	13.2	0.24	28.9	71.2 (***)
100-200'	7.1	13.8	0.01	28.9	72.4 (***)
200-400'	7.1	14.3	0.18	28.9	74.7 (***)
400-1000'	7.1	13.4	0.18	28.9	71.5 (***)
>1000'	7.1	14.7	0.42	28.9	76.7 (***)
Winter					
<100'	0.2	2.2	12.07	37.9	38.3 (*)
100-200'	0.2	0.2	11.08	37.9	32.3 (NS)
200-400'	0.2	3.2	7.42	37.9	28.6 (NS)
400-1000'	0.2	5.8	5.00	37.9	29.2 (NS)
>1000'	0.2	0.8	1.25	37.9	6.0 (NS)

Table 2-3. Relationship between land use/cover variables (UR/AG = urban/agriculture; SQMI = watershed area; FR/AG = forest/agriculture area) and mean summer nitrate-N and reactive phosphorus concentrations (mg/L) generated from multiple regression analyses with respect to buffer regions. The SQMI variable is in square miles; UR, FR, and AG are in acres.

Buffer <100 ft

Mean NO_3-N = 0.818 + 0.00278SQMI + 33.5UR/AG - 2.2FR/AG
Mean reactive P = 0.079 + 0.00700SQMI + 45.5UR/AG - 1.8FR/AG

Buffer = 100-200 ft

Mean NO_3-N = 0.742 + 0.00247SQMI + 29.8UR/AG + 0.6FR/AG
Mean reactive P = 0.050 + 0.00838SQMI + 41.8UR/AG - 2.2FR/AG

Buffer = 200-400 ft

Mean NO_3-N = 0.776 + 0.00308SQMI + 31.1UR/AG - 8.7FR/AG
Mean reactive P = 0.045 + 0.00788SQMI + 42.0UR/AG - 7.7FR/AG

Buffer = 400-1,000 ft

Mean NO_3-N = 0.734 + 0.00328SQMI + 30.8UR/AG - 30.1FR/AG
Mean reactive P = 0.011 + 0.00840SQMI + 40.8UR/AG - 30.8FR/AG

Buffer > 1,000 ft

Mean NO_3-N = 0.584 + 0.00316SQMI + 8.1UR/AG + 49.9FR/AG
Mean reactive P = 0.248 + 0.00836SQMI + 11.0UR/AG + 85.8FR/AG

Table 2-4. Example of screen output from interactive session examining a proposed 100-acre conversion of agriculture land to urban land and the predicted effects on instream nitrate-N and soluble reactive phosphorus concentrations in the northwest portion of the Salt Fork watershed.

```
SETUP
AGRICULTURAL ACRES?  28240.
FOREST ACRES?  15.
URBAN ACRES?  4086.
LAKE ACRES?  3.
BARREN ACRES?  83.

CHANNEL LOCATION (SQR MILES)?  48.
OK

RESET OK
```

INTERACTIVE LAND-USE IMPACT ANALYSIS

SYSTEM VERSION 1.0

	CURRENT	PROPOSED
(landuse in acres)		
AGRICULTURE	28240.00	28240.00
URBAN	4086.00	4086.00
FOREST	15.00	15.00
LAKES	3.00	3.00
BARREN	83.00	83.00
(chemical conc. mg/l)		
ORTHO-PO_4	3.40	3.40
NO_3	3.17	3.17

at 48.00 sq miles drainage

```
TRANSFER from?  AG
to?  URBAN
acres?  100
AVERAGE FEET FROM CHANNEL?
400.  OK
```

INTERACTIVE LAND-USE IMPACT ANALYSIS

SYSTEM VERSION 1.0

	CURRENT	PROPOSED
(landuse in acres)		
AGRICULTURE	28240.00	28140.00
URBAN	4086.00	4186.00
FOREST	15.00	15.00
LAKES	3.00	3.00
BARREN	83.00	83.00
(chemical conc. mg/l)		
ORTHO-PO_4	3.40	5.72
NO_3	3.17	4.85

at 48.00 sq miles drainage

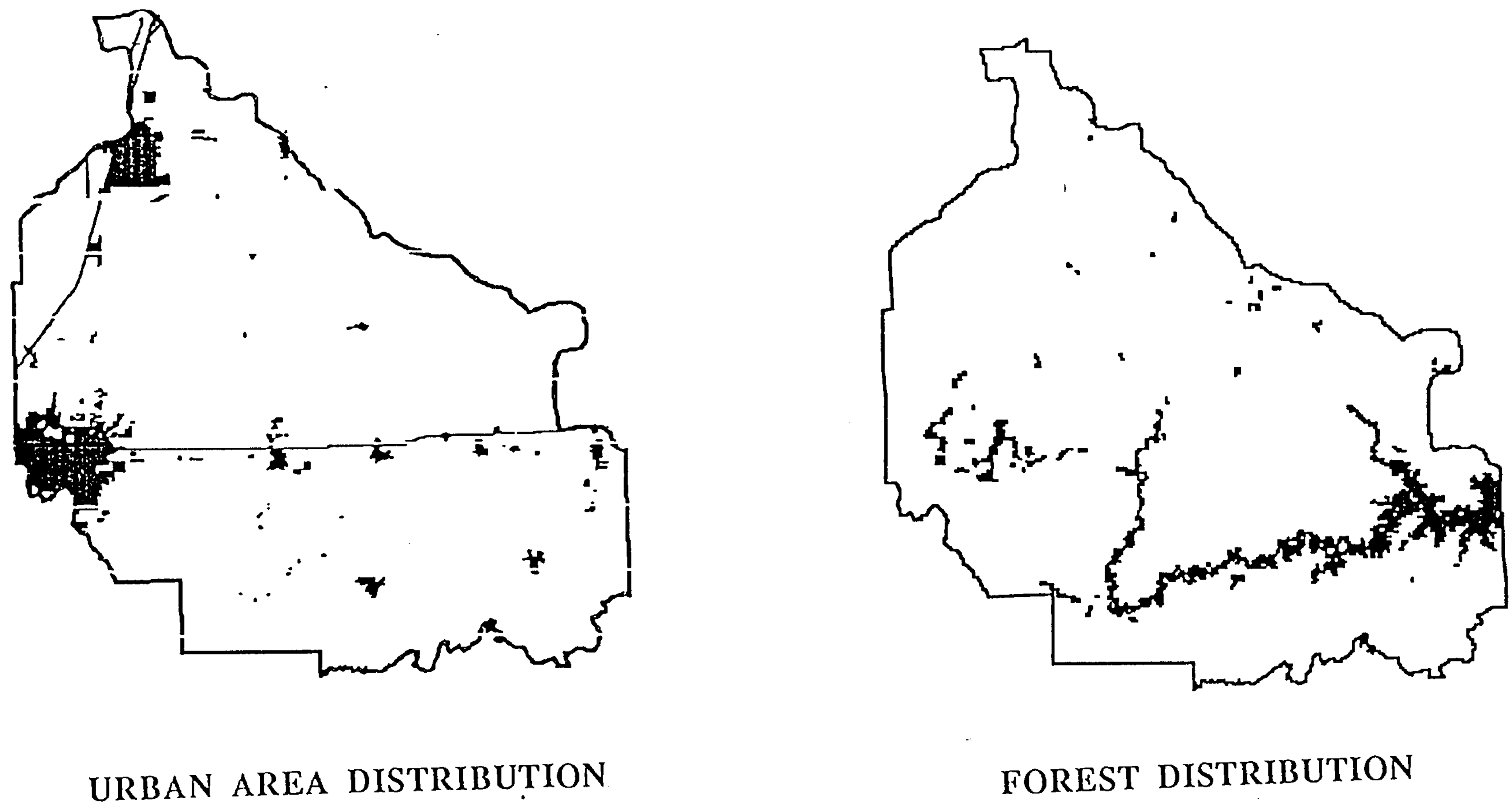

Fig. 2-1. The Salt Fork watershed and the distribution of urban and forest areas. More than 99% of the remaining non-black areas were in row crop agriculture.

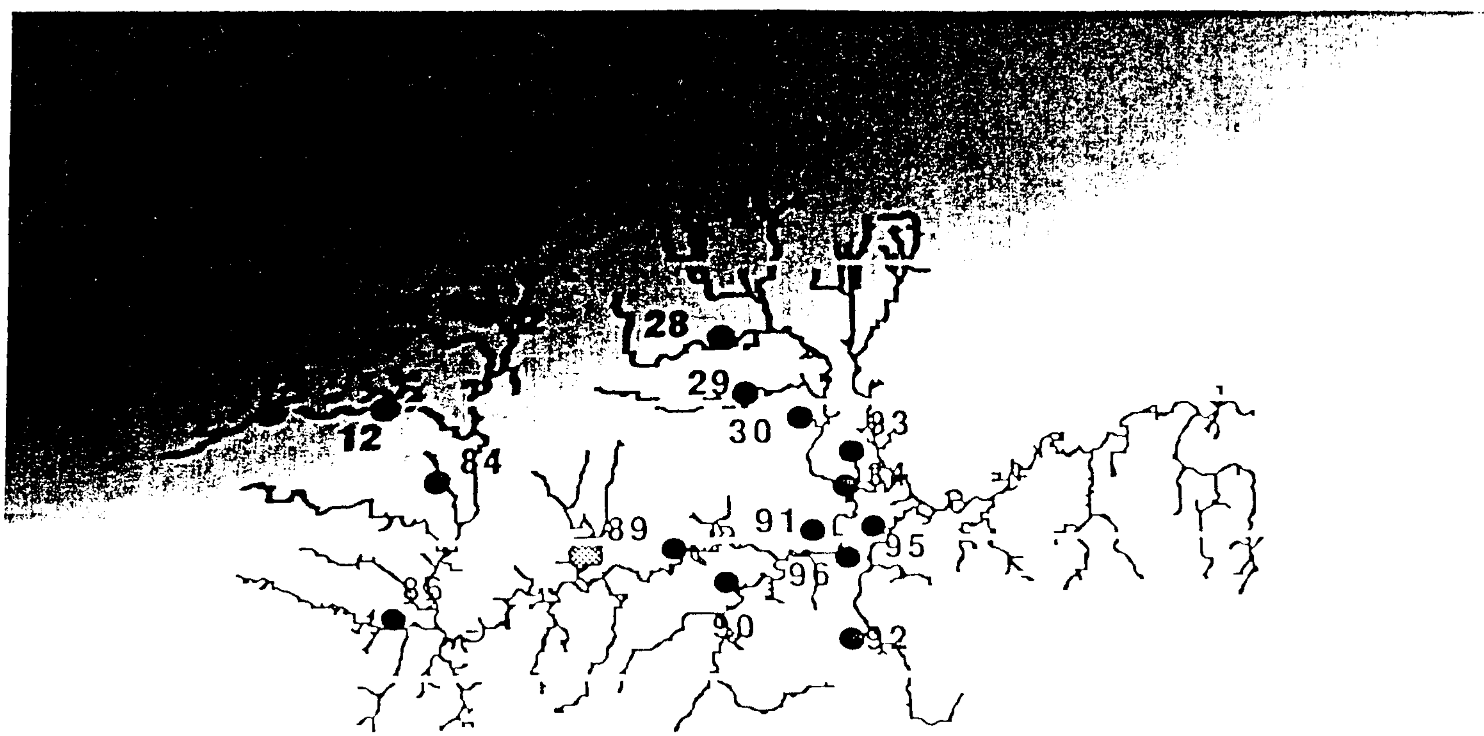

Fig. 2-2. Stream hydrology and location of chemical sampling stations for the Salt Fork watershed (scale 1:250,000).

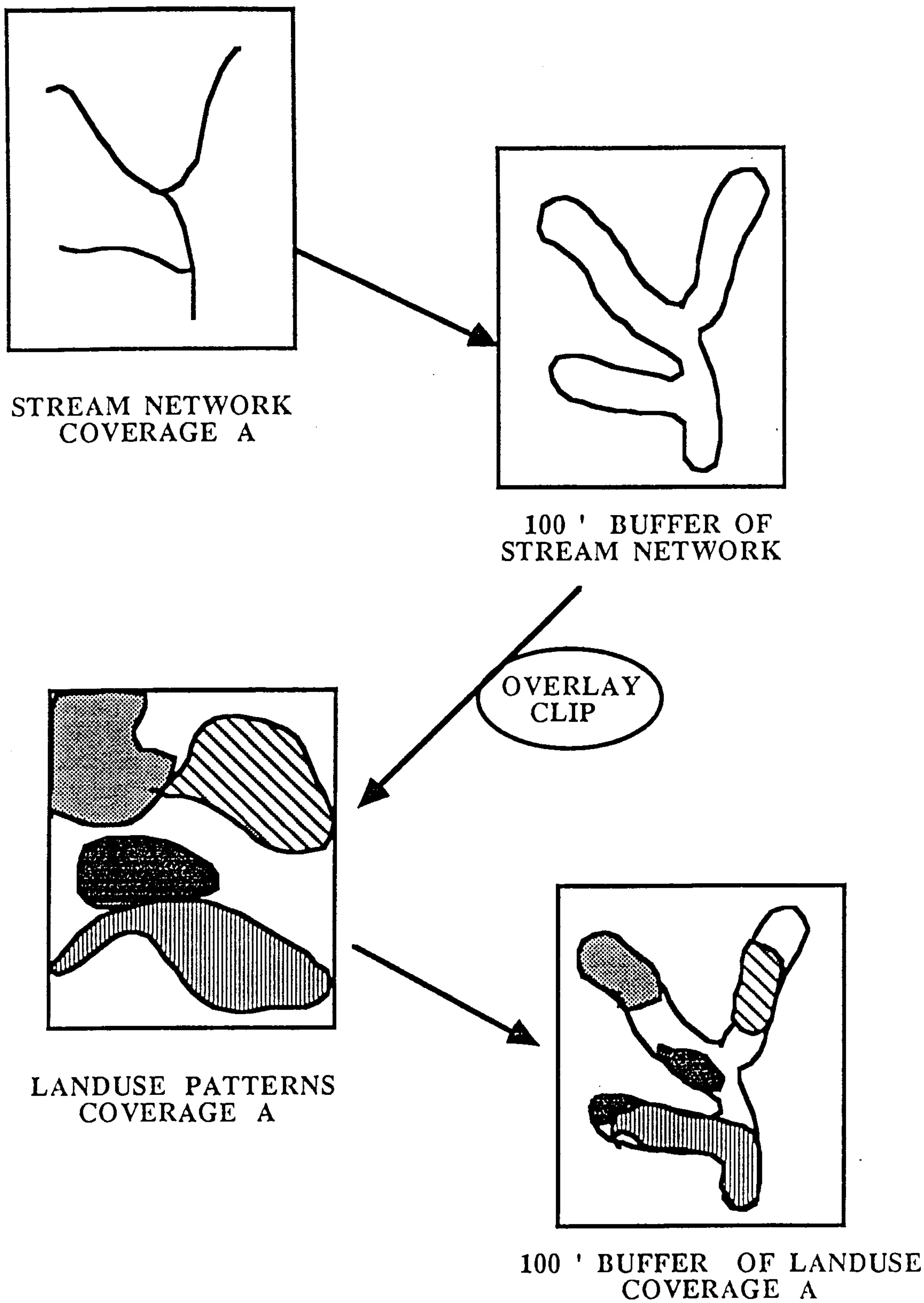

Fig. 2-3. Arc/Info buffering facility employed in analyses.

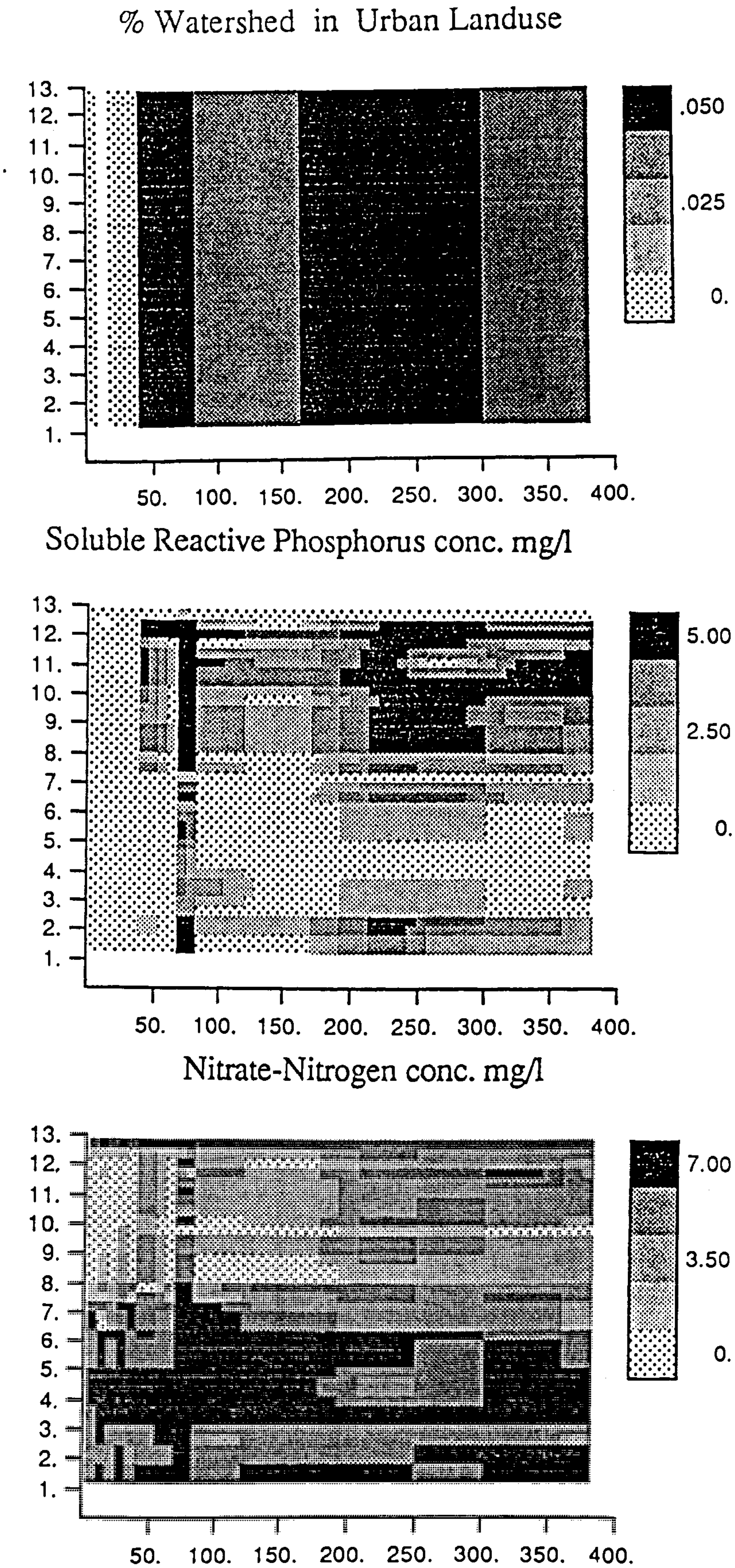

Fig. 2-4. Monthly (y-axis) concentrations (mg/L) of soluble reactive phosphorus and nitrate-N in the Salt Fork watershed with respect to sampling station drainage area (x-axis; square miles) and the percent distribution of urban areas with respect to drainage area.

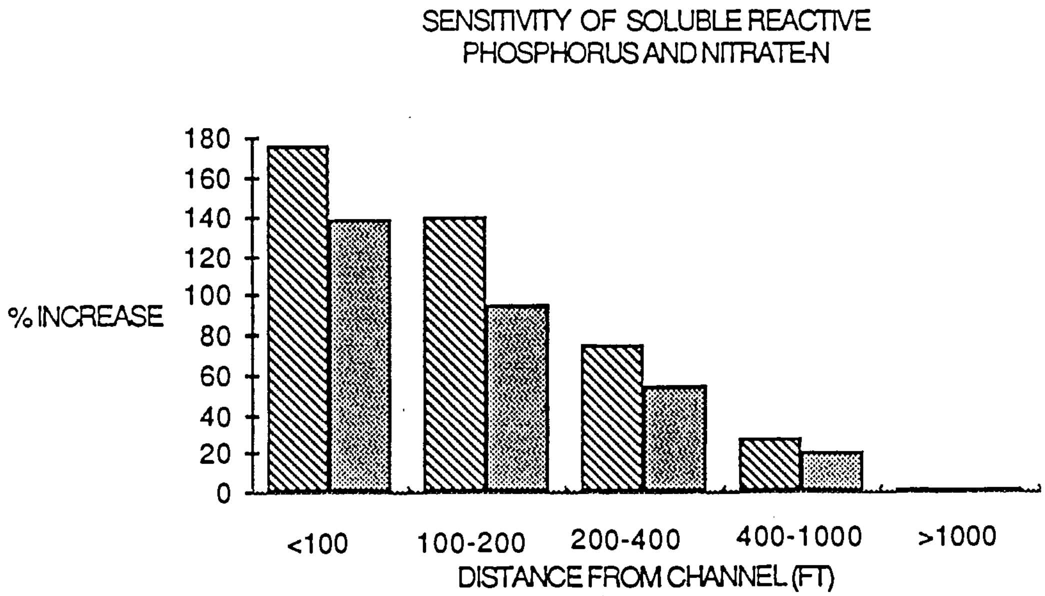

Fig. 2-5. The predicted percent change in the concentration of soluble reactive phosphorus (diagonal hashes) and nitrate-N (dotted bars) in the Salt Fork following the conversion of 100 acres of agriculture land to urban land use within each watershed buffer region.

Chapter 3

Large-scale Patterns of Nutrient Availability and Primary Production in an Agriculturalized Prairie River System

INTRODUCTION

The influence of streamside vegetation on the structure and function of stream ecosystems has received growing attention over the past decade (Cummins 1973, 1974; Cummins et al. 1984; Hynes 1975; Vannote et al. 1980). Recently, riparian vegetation has been proposed as the primary organizing factor in the theoretical constructs arising out of the river continuum paradigm (Cummins et al. 1984, Minshall et al. 1985). It is surprising, then, that grasslands in general, and the agriculturalized prairie watersheds of North America in particular, have received so little attention. Grasslands and agricultural fields cover roughly 42% of the earth's land surface (Smith 1974), and watersheds draining these areas provide an almost ideal inversion of the normal longitudinal structure of riparian vegetation described in the continuum literature (Vannote et al. 1980). For this reason, we believe that prairie watersheds provide an important natural experiment in which theories about the impact of patterns in riparian vegetation (and other large-scale watershed characteristics) on stream ecosystems can be empirically tested and refined. In this respect, prairie streams are similar to those of more arid desert regions (Busch and Fisher 1981, Grimm et al. 1981, Cushing and Wolf 1984, Grimm and Fisher 1986). However, unlike true desert systems, prairie rivers have perennial flows and are hydrologically a more useful contrast to rivers in forested uplands.

Previous investigations of the metabolism of prairie river systems have demonstrated that upstream sites are frequently, but not necessarily, autotrophic. Duffer and Dorris (1966) reported gross primary production rates of up to 48 g O_2 $m^{-2} \cdot d^{-1}$ in Oklahoma but found only one of three sites consistently had a P/R ratio greater than one. Gross primary productivity reached a maximum of 32 g O_2 $m^{-2} \cdot d^{-1}$ in an agricultural watershed in Kansas and mean P/R ratios exceeded one in half the sites examined (Prophet and Ransom 1974). Gelroth and Marzolf (1978) found a P/R ratio greater than two in a small stream on the Konza Prairie. High nutrient levels (N and P) are also characteristic of prairie watersheds, in fact often are directly attributed to agricultural activities (Prophet and Ransom 1974, Kilkus et al. 1975, Omernick 1977).

In this chapter, we summarize the results of a 2-yr investigation of spatial and temporal patterns in community metabolism in a typical prairie watershed in east-central Illinois. Gross primary productivity (GPP), whole community respiration (CR), nutrient concentrations (soluble reactive phosphorus and nitrate-N) and physical control variables (percent shade, temperature, turbidity, and land use) were examined in relation to the longitudinal distribution of riparian vegetation in this heavily agricultural watershed. Multiple linear regression techniques were used to derive empirical models for photosynthesis and community respiration. We argue that the longitudinal distribution of photosynthesis principally reflects downstream patterns in light limitation and not nutrient availability in this river system. The utility of the river continuum concept (RCC) (Vannote et al. 1980) in understanding the structure and function of agriculturalized prairie watersheds is discussed.

METHODS

Study Area

This study was conducted in two branches of the Vermilion River, a tributary stream of the Wabash, located in east-central Illinois. The watersheds of the Salt Fork and the Middle Fork are adjacent and encompass approximately 465 and 428 miles^2, respectively. Thirty-two sampling locations were established in the two watersheds (Fig. 3-1a) and were sampled periodically over 2 years for physico-chemical characteristics and rates of community metabolism. The upper regions of both watersheds were channelized to facilitate drainage from 1890 to 1920. No significant new channelization has occurred within the study area during the past 60 years. The early channelization had the effect of increasing the total length of the drainage network by extending upstream channels into prairie wetlands. This, in turn, has likely resulted in more variable hydrographs throughout the watershed. Substrate composition in both rivers is similar. Substrate in the upper portions consists of coarse sands, reflecting the low slope of these areas. Corresponding with increases in slope, lower reaches of both rivers are dominated by small cobble and rubble in riffle areas and by sand and silt in pools.

Riparian Structure and Land Use

Watershed boundaries for each sampling station were digitized from 7.5-min (1:24,000) U.S.G.S. topographic maps. Land-use/cover patterns were determined for each station watershed by interpreting false-colored infrared National High Altitude aerial photographs (1:20,000 scale; film exposure April 1981 and 1983; EROS Data Center, U.S. Dept. of Interior, Sioux Falls, SD). All data were interpreted onto mylar sheets to minimize distortion, digitized, and analyzed using ARC/INFO (Environmental Systems Research Institute), a geographic information system running on a PRIME 750 mini-computer. Six land-use classifications were employed in the study: (a) urban and developed lands, (b) agricultural land, (c) forested land, (d) water (lakes and streams), (e) wetlands, and (f) barren lands (*e.g.*, mining areas). Land use was characterized for each station as the proportional contribution of each category to the total drainage area.

Detailed site inventories were made at each station; mean slope of the site; mean substrate particle size; variation in substrate particle size; percentage of sand, cobble, and rubble; percentage of channel shaded (by riparian vegetation or bank height); base flow mean depth; coefficient of variation in depth across a transect; mean velocity; and hydraulic diversity were measured. Mean slope was determined from 7.5-min topographic maps. Substrate data were obtained once from each station by collecting sediments along 6-9 transects perpendicular to the channel flow in a three riffle-three pool sequence. The percentage of channel shaded was a composite estimate obtained by walking 500 m upstream along the stream bank and visually estimating vegetation canopy across the channel width.

Water Chemistry

Water chemistry data were collected at approximately 2-week intervals from December 1983 to December 1985. Samples were collected during both storm and non-storm events, providing representative water chemistry concentrations during the study period. Nitrate-N, nitrite-N, ammonia-N, turbidity, minimum and maximum biweekly temperature, specific conductance, pH, and soluble reactive phosphorus (SRP) were measured. Water temperatures were measured using mercury-filled, magnet-reset maximum-minimum thermometers anchored in the stream.

Duplicate water samples were collected in acid-rinsed (1:1 HCl) 500-ml glass bottles and returned to the laboratory for analysis. One sample from each station was preserved with 1.5 ml of sulfuric acid (5.25 N) to adjust the sample to a pH of 2. Prior to analysis, water samples were returned to room temperature and the preserved sample was adjusted to a pH of 7.0 by adding a standard solution of 5 N sodium hydroxide. Subsamples of the unpreserved water sample were used to determine specific conductance, turbidity, pH, nitrate-N, and nitrite-N. Specific conductance (µohms/cm) was measured using a Yellow Spring Model 33 salinity-conductivity meter, turbidity (NTU) with a Monitek Model 21 laboratory nephelometer, and pH using a Sargent-Welch TBX pH meter or a Cole Palmer Model 5983 chemcadet.

Nitrate-N was measured using the low-range cadmium reduction method or the high-range cadmium reduction method substituting gentisic acid for 1-naphthylamine. Nitrite-N was determined using the diazotization procedure incorporating chromotropic and sulfanilic acids as indicators. Ammonia-N was measured using the Nessler method (Hach Chemical Co.). Soluble reactive phosphorus was determined using a modification of the molybdenum blue procedure.

Quality control tests were conducted in March 1984 to assure the accuracy of the above procedures and the precision of the laboratory analysts. Voluette ampule standards with known concentrations of nitrates, ammonia, and reactive phosphorus were supplied by the U.S. Environmental Protection Agency. Two replicates of each test were performed on voluettes with varying concentrations of each nutrient. Voluette concentrations were unknown to laboratory analysts prior to and during testing. All results fell within the 90% confidence interval, and all SRP and ammonia-N results were within the 95% confidence interval.

Community Metabolism

Community photosynthetic and respiration rates were estimated using a single-station open-channel regression technique (Owens 1974, Kosinski 1984) based on Odum's original single-station method (Odum 1956). Diel records of oxygen concentration and temperature (and in some cases, light) were obtained from selected stations using automated data logging units (n = 191) or manual monitoring (n = 97). In both cases, we employed saturation-calibrated YSI Model 57 oxygen meters. Meter drift during automated collection was generally <1 ppm and was distributed evenly throughout the data set. The mean night-time respiration rate and reaeration coefficient (K) were estimated from each data set by least-squares regression of the after-dark data:

$$\Delta C/\Delta t = R + K D$$

where C is oxygen concentration in mg/L; t is time (h); R is mean respiration rate (mg O_2 $L^{-1} \cdot h^{-1}$); K is the reaeration coefficient (1/t) ; and D is the oxygen deficit (O_2 at saturation - observed O_2). Gross community photosynthesis was then estimated per quarter-hour of daylight from the relation:

$$GPP_h = (\Delta C/\Delta t) + R - K D$$

where GPP_h is gross community photosynthesis in mg O_2 $L^{-1} \cdot h^{-1}$. Daily GPP was estimated as mean GPP_h x hours of daylight. Daily community respiration (CR) was estimated as 24 x R.

RESULTS

Vegetation and Land-use Structure

The Salt Fork and the Middle Fork watersheds are both heavily agriculturalized, with 91 and 93% of their drainage areas, respectively, in intensive rowcrop production (corn and soybeans) (Table 3-1). However, the two watersheds differed substantially in extent of urbanization (Fig. 3-1b). Seven urban (primarily residential) areas are located in the Salt Fork watershed, including the cities of Urbana and Rantoul and the towns of Sidney, St. Joseph, Homer, Muncie, and Oakwood. Only one urbanized area of any size, Paxton, is located in the drainage basin of the Middle Fork. Population centers on the Salt Fork contributed to an overall density of 138 individuals/ mile^2, as opposed to a density of 15/mile^2 for the Middle Fork basin. Despite differences in absolute urbanization, the longitudinal distribution of urban land use was similar in both watersheds, with the highest relative proportional contributions occurring at stations with a drainage of 200-300 miles^2.

Significant riparian vegetation is restricted to mature gallery forests on the lower portions of the mainstem in both rivers (Fig. 3-1c). Comparisons of present forest distributions with presettlement distributions as mapped by 19th century cartographers indicate that, despite a significant narrowing of the gallery forests due to agricultural encroachment, the longitudinal extent of the gallery forests has remained remarkably consistent during the past 140 years. The original extent of the gallery forests are clearly reflected in the present-day distributions of soil associations within the watersheds (Fig. 3-1d). Comparisons of the distributions of forest-derived soils and modern woodlands suggest that deforestation since white settlement has reduced total forest cover by approximately 72% in the Middle Fork and 74% in the Salt Fork. Presently, and we believe historically, both percentages of the watershed forested and of the channel shaded tend to increase in a downstream direction and are significantly correlated with the logarithm of drainage area ($r = 0.367$, $P < 0.05$ and $r = 0.548$, $P < 0.01$, respectively). Some small channelized tributary streams (drainage area < 10 $miles^2$) are significantly shaded by the channel wall itself, although this is mainly an artifact of early channelization.

Thermal Structure and Turbidity

The Vermilion River drains relatively impermeable soils; as a result, both watersheds receive most of their water from surface and shallow subsurface (field tile) runoff. The small contribution of cold groundwater to total discharge, combined with a lack of shading in upstream reaches, results in a relatively homogeneous average temperature throughout the entire watershed at any given time (correlation with drainage area = 0.059, $P > 0.05$). However, observed diel ranges in temperature were highly dependent upon a site's position in the watershed. Daily maximum temperature declined in the downstream direction and was negatively correlated with drainage area ($r = -0.2$, $n = 204$, $P < 0.01$), but daily minimum temperature increased with drainage area ($r = 0.26$, $n = 205$, $P < 0.01$). Diel temperature fluctuations, therefore, decreased with increasing stream size (*i.e.*, thermal stability increased with increasing flow volume). Temperature variability was highest in the spring and fall with diel ranges sometimes in excess of 15°C. But even during the summer, diel fluctuations of 10°C were frequently observed in the upper parts of the Vermilion River watershed.

Turbidity in the Vermilion River, as in most Midwestern rivers, was generally high and variable. Turbidity in biweekly sampling ranged from <1 to >600 NTU. Turbidity increased with increasing discharge ($r = 0.53$, $P < 0.0001$) and was related statistically both to increased suspended loads associated with high flow events (flows with low exceedence frequency) and to a tendency of increasing turbidity in the downstream direction (Table 3-2). The Middle Fork watershed was significantly more turbid than the Salt Fork ($T = 5.714$, $P < 0.001$), particularly during high flows, because of the larger proportion of clayey soils in the upper reaches of its watershed.

Nutrient Availability

Nutrient concentrations were extremely high, but there was a large amount of variability between seasons and between sites (Table 3-3). Total inorganic nitrogen concentrations ranged from 0.323 to 12.7 mg/L, with an annual mean concentration for both watersheds of 4.8 mg/L. Soluble reactive phosphorus (SRP) ranged from 0.003 to 15.2 mg/L and averaged 0.6 and 1.5 mg/L in the Middle Fork and Salt Fork, respectively. The large-scale distribution of nutrient availability within these watersheds was primarily controlled by land-use patterns, principally urbanization in the case of SRP, and both urbanization and seasonal field applications by farmers in the case of nitrate (Fig. 3-2). Nitrate concentrations were generally similar in both watersheds with concentrations >5.0 ppm not uncommon. During summer and fall, the highest concentrations of nitrate were associated with urban areas (appearing as vertical streaks in Figs. 3-2a and 3-2b), representing point-source inputs. Concentrations in the Salt Fork were generally higher than those in the Middle Fork during these periods, due to greater urbanization. However, from late fall through the high-water period in late spring-early summer, nitrate was extremely high throughout both watersheds. The second period coincided with agricultural ammonia applications, which begin in November-December, peak in May, and are completed by June.

SRP concentrations were highly correlated with urbanization. During low-flow periods in the Salt Fork, concentrations downstream of urban effluent inputs routinely exceeded 5 ppm SRP; these reaches appear as vertical streaks in the SRP watershed profile (Figs. 3-2a and 3-3b). During low-flow periods, high concentrations in the SRP profile tend to "bleed" downstream. During high flow, this longitudinal patterning is diluted. The Middle Fork basin had little urbanization and relatively low SRP concentrations; although SRP exceeded 5 ppm at times in the vicinity of the only population center, concentrations generally ranged from 0.2 to 1 ppm, low by Vermilion River standards but still high in comparison with many forested and desert streams in North America (Omernick 1977).

Community Metabolism

Rates of gross primary productivity varied from below detection limits to >50 g O_2 $m^{-2} \cdot d^{-1}$ (>15 g carbon $m^{-2} \cdot d^{-1}$). Rates were highest during summer and in the upper portions of the watershed (Figs. 3-3 and 3-4). Multiple regression analyses indicated that water temperature and light availability (as mediated by turbidity and shading) were the primary variables controlling rates of primary production (Table 3-4). A combination of temperature, turbidity, depth (depth x turbidity is proportional to extinction coefficient), and maximum percentages of the channel shaded explained 61% of the variance in 195 observations across all months, over 30 sites, and two watersheds. When data from only the summer growing season were used in the analysis, temperature was not significant, and turbidity and shading explained most of the variation in GPP ($r^2 = 0.77$, $P < 0.0001$) (Table 3-5). Neither N nor SRP concentrations (nor their transformations) could be correlated with the residuals of these fits, suggesting that nutrient limitation in this watershed is unimportant. In exploratory single-station analyses, nitrate concentrations were found to be negatively correlated over time with GPP at a few highly photosyn- thetic sites. This likely reflects high rates of nutrient removal but does not necessarily indicate nitrate limitation. No significant difference between the Salt Fork and Middle Fork basins was found, unless turbidity, which was higher in the Middle Fork, was removed from the regression. In both analyses, the relationship of each key variable to GPP was best described as a power function of the form:

$$\text{GPP per unit volume} = a\, T^{+b}\, NTU^{-c}\, S^{-d}\, H^{-e} \qquad (1)$$

where GPP is gross primary productivity (g O_2 $m^{-3} \cdot d^{-1}$); T is temperature (°C); NTU is turbidity (nephelometric units); S is maximum percentage of channel shaded; and H is mean (hydraulic) depth (m); and a, b, c, d, and e are coefficients of the indicated sign.

Because the exponent for depth (e) is negative but greater than -1 (-0.68 and -0.45 from Tables 3-4 and 3-5), GPP on a per unit channel volume basis declines as mean depth increases in a downstream direction. If the expression is converted to the more familiar GPP per unit area (by multiplying by mean depth), the exponent is positive but less than 1. For example, the exponent for depth from Table 3-4 is -0.82; multiplying by mean depth is equivalent to adding 1.0 to this exponent, which yields a value of 0.18. On a unit area basis then, GPP should increase in a downstream direction, rapidly at first and then more slowly as the slope of the power function asymptotically approaches zero.

The seasonal pattern in photosynthetic production was similar in both watersheds. In spring, GPP rose first in the upstream reaches with drainage areas of 20-100 miles^2. By June, GPP exceeded 10 g O_2 $m^{2\cdot}$ d^{-1} in almost all first -and second-order streams, and by August high levels of photosynthesis could be found throughout the watershed. As water temperatures declined in the fall, rates of GPP declined, first in the upstream reaches and finally to <2 g O_2 $m^{-2\cdot}$ d^{-1} throughout the entire watershed by December.

Respiration rates ranged from undetectable to 177 g O_2 $m^{-2\cdot}$ d^{-1} and were highly correlated with photosynthetic rates ($r = 0.534$, $n = 218$, $P < 0.001$, log transformed). Multiple regression analyses of respiration data were less successful than those for photosynthetic rate data. Although r^2 values were generally lower than those in the GPP regression, statistically significant models of biologically reasonable form (Tables 3-6 and 3-7) were generated:

$$\text{CR per unit volume} = a\,(\text{GPP} + 1)^b\,T^c \qquad (2)$$

where CR is community respiration (g O_2 $m^{-3\cdot}$ d^{-1}). As in the case of photosynthesis, when the analysis was based on mean summer rates, the temperature effect was no longer significant but the fit was substantially improved ($r^2 = 0.69$, $P < 0.0001$; Table 3-7). Conversion of Eq. 2 to a unit area basis suggests that the respiration rate per unit area increases as a linear function of depth in these watersheds.

The ratio of GPP to CR (*i.e.*, P:R ratio), an index of relative autotrophy, was highly variable between seasons and between sites. Generally, the ratio was correlated with GPP ($r = 0.725$, $n = 174$, $P < 0.001$, log transformed). During the summer (July-September), 60% of the sites examined had a P:R ratio < 1, while 40% could be classified as autotrophic. This percentage closely approximates the relative frequency of autotrophy in all summer observations and is higher than the frequency of autotrophy when observations from all seasons were pooled (31%). There was a clear longitudinal pattern in the GPP:CR ratios, with most autotrophic sites occurring high in the watershed (Fig. 3-5). This observation is consistent with the expected distribution using Eqs. 1 and 2, because the ratio of depth dependencies for GPP and CR per unit area is a power function with a negative exponent. However, the variances in these data were high, and low ratios were not uncommon in the upper watershed. This variation indicates that the other controlling variables (temperature, shading, and turbidity) can and do vary independently of depth.

DISCUSSION

Nutrient levels in the Vermilion River were surprisingly high, even for an agricultural stream in this region. Omernik's (1977) survey of land-use relationships to instream water quality suggests that mean annual concentrations of total inorganic nitrogen usually range from 2.87 to 6.64 mg/L for Midwestern streams with 100% of their watersheds in agricultural and urban land use. The annual mean concentration in the Vermilion River of 4.8 mg/L is within this range. Phosphorus concentrations, on the other hand, are substantially higher than Omernik's (1977) predicted range of 0.073-0.251 mg/L for total P. SRP concentrations alone averaged 0.6 and 1.5 mg/L in the two river basins examined, with maximum concentrations exceeding 5 mg/L. Agricultural activity is usually cited as the primary cause of high nutrient conditions in streams in the "corn belt" (Kilkus et al. 1974, Omernik 1977, Goldman and Horne 1983). Soil erosion, in particular, commonly has been identified as the major contributor of phosphorus to agricultural watersheds. Although soil-borne phosphorus (usually organic) contributes large portions of the annual transported load in many streams, most of this transport takes place over very short intervals following major hydrologic events (Baker 1984). Data from the Vermilion River suggest, however, that for average instream conditions urban, not agricultural, land-use patterns control the longitudinal distribution of SRP concentrations. Urban areas (point and non-point sources are indistinguishable) clearly dominate the watershed profiles for SRP and for nitrate during most of the growing season, elevating concentrations 2-5 times the background levels (Fig. 3-2). Agricultural inputs of nitrogen were detectable and clearly controlled instream concentrations during late fall-early spring, corresponding to periods of very low temperature and/or high water levels and minimal biological activity.

Factors Limiting Primary Production

Although gross photosynthetic rates in the Vermilion River were variable, observed maximums were as high or higher than previously reported values for agriculturalized grasslands biomes (Table 3-8). Streams draining temperate grasslands tend to be more productive than forested or more arid watersheds. This productivity can be attributed to a combination of factors, the most important of which is that their shallow, warm headwaters are open to light. Furthermore, as is clearly the case in the Vermilion River, these streams tend to be rich in phosphorus and nitrogen.

While photosynthetic rates were highest in the smaller headwater streams, not all low-order streams were highly productive. This variability in photosynthetic production between sites of a similar drainage area was largely due to differences in shading. There was a striking difference between photosynthetic rates of reaches high in the drainage network and of main-stem tributaries of the same order which entered the main river valley passing through its associated gallery forests. For example, station 95 (second order) was located on a small tributary to the main stem of the Vermilion River and had a drainage of 21 miles^2. Because the stream flows into the Salt Fork at a point where it is a fifth-order river, much of its lower reach is enveloped in extensive gallery forest, and this site was consequently heavily shaded. The mean summer photosynthetic rate at station 95 was 11.7 g O_2 $m^{-3} \cdot d^{-1}$. Station 18 had a similar-sized watershed (24 miles^2, second order), but it was located in the upper portion of the Salt Fork drainage network, removed from the influences of the gallery forests. At this site the mean summer rate of photosynthesis was 29.7 g O_2 $m^{-3} \cdot d^{-1}$, almost three times higher. Thus, it is not possible to accurately characterize the relationship between stream size (drainage area or stream order) and expected rates of primary production without first specifying assumptions about the availability of light.

In the Vermilion River drainage, nutrient availability appeared to have little importance in controlling the distribution of photosynthetic rates. No significant positive correlations were found between GPP and nitrate-N or SRP concentrations, either on an annual basis or when data from only summer (June-September) were examined. Recent nutrient addition experiments (Munn et al., personal communication) at seven sites in these watersheds also support the hypothesis of no nutrient limitation. Light availability, as mediated by turbidity, depth, and shading, appears to be the primary limiting factor that controls the large-scale pattern of photosynthesis in the Vermilion River.

Several factors interact and together control the amount of radiant energy available for instream photosynthesis. Shading by riparian vegetation has been most often identified as the key light-limiting factor in stream ecosystems (Vannote et al. 1980, Cummins et al. 1984). However, in the Vermilion River, as presumably in most turbid rivers, light extinction due to scattering is equally important. The amount of light extinction that occurs is a function of both the light scattering capacity of water (turbidity) and the distance light must travel in water (depth). In limnological investigations, light extinction has historically been measured empirically and described in terms of an exponential decay ($\partial L/\partial$ depth = -k · depth). The decay coefficient, k, is then used as an index of light extinction. Nephelometric turbidity (NTU) is a routinely measured parameter in water quality surveys that can be used as a functional analog of the extinction coefficient. Light availability in our empirical model for GPP is represented by the composite term $S^bH^cNTU^d$. This term encompasses three independent parameters that control the amount of light which can penetrate to periphyton communities on the channel bottom: shading, depth, and turbidity. Examination of the partial sums of squares from the regressions (Tables 3-4 and 3-5) gives some indication of the relative importance of these three terms in the Vermilion River. In the analysis of the entire data set (Table 3-4), the shading variable adds very little to the overall regression, suggesting that, on an annual basis, a very large portion of the variance in photosynthesis is related to seasonal cycles in temperature and high water. If mean summer GPP values for each station are used, and the analysis restricted to the summer growing season when shading effects are at a maximum and turbidity at a minimum (Table 3-5), the shading variable and turbidity effects (depth and NTU) are roughly equivalent (partial F's = 13.3, 7.3, and 3.4 for shading, turbidity, and depth, respectively). Thus, we conclude that overall turbidity is at least as important as riparian shading in limiting light availability to this particular system.

Prairie Watersheds and the River Continuum Concept

Clearly the large-scale structure of the Vermilion River watershed is different from the typical structure of forested watersheds. In terms of riparian vegetation, temperature and community metabolism (GPP and CR), longitudinal distributions are basically inverted (Fig. 3-6).

The upper portions of the watershed, which drain agriculturalized prairie, are open and unforested and have been so prior to white american settlement in the 1840s. Gallery forests and attendant leaf CPOM inputs eventually develop along the drainage network but are restricted to the lower half of the watershed. This situation is exactly the reverse of the "prototype" configuration posited by Cummins et al. (1984) for non-xeric river systems. Open headwaters are more typical of xeric regions like deserts, high altitudes, *etc.* (Cummins et al. 1984, Busch and Fisher 1981). Prairies and other grassland ecotypes are mesoxeric systems and owe their peculiar vegetational structure to this fact and to historically high disturbance regimes.

High ratios of groundwater to run-off are common in upland systems and usually result in marked downstream temperature gradients (Vannote et al. 1980, Minshall et al. 1985), with diel variability increasing in the downstream direction. In the Vermilion River, groundwater flow is relatively low and there was very little longitudinal variation in median temperature. Diel variability, in the absence of the moderating influence of groundwater, declined with increasing drainage area. Without a riparian canopy, and coupled with low flow volumes (reduced thermal capacitance), upstream sites tended to move quickly towards a thermal equilibrium with air temperature, mimicking diel fluctuations in the terrestrial environment. Downstream, larger thermal capacitance is augmented by insulating tendencies of the riparian canopy; together, they damp the thermal response of the river to diel changes in radiation inputs. Thus, we conclude that differences between the thermal characteristics of this river and those of typical boreal rivers result both from regional characteristics of the hydrology and from the unique structuring of riparian vegetation in prairie watersheds.

Given the absence of shading vegetation and higher daytime temperatures, it is not surprising that we found the highest rates of autotrophic production in the upper reaches of this watershed. Forested upland watersheds are usually described as being heavily heterotrophic in their upper reaches, with autotrophic activity increasing in mid-order streams where the river is wide enough to break the canopy cover and admit more light to the water's surface (Cummins 1974, Minshall 1978, Vannote et al. 1980, Minshall et al. 1985). Declines in photosynthetic activity occur further downstream because of light attenuation due to increasing depth and turbidity. In the Vermilion River, the most photosynthetically active sites occurred at sites with drainage areas of <100 miles^2 (first-third order). Beyond that point, a combination of depth, turbidity, and forest canopy reduce light inputs significantly and result in a corresponding decline in GPP per unit stream area. Although channel shading began to wain beyond 300 miles^2, no significant increase in photosynthetic activity was detected, presumably because of light extinction due to turbidity and increased depth.

In its simplest form, the RCC posits a pristine structure for river ecosystems that begin with heterotrophic headwaters dominated by allochthonous inputs from riparian vegetation, grades into an autotrophic middle section, and ends in a large river that is again primarily heterotrophic (Vannote et al. 1980). In the Vermilion River, the longitudinal patterning of community metabolism is distinctly different. However, recent elaborations of the RCC (Cummins et al. 1984, Minshall et al. 1985) have more flexibly addressed the relationship between stream metabolism and patterns in riparian vegetation. While a continuum of dominant energy sources is still proposed (allochthonous production to autochthonous production to lateral and upstream accrual), individual streams are considered to enter this continuum at different points; xeric systems are described as entering the continuum below the zone of shading and allochthonous input from riparian forests (Minshall et al. 1985). Desert rivers, then, are viewed as having high P/R ratios in their upstream reaches, with instream periphyton production being replaced first by macrophyte production and finally by floodplain and upstream accrual. This expanded version of the RCC, by stressing the controlling role of riparian vegetation, could be said to predict the upstream autotrophy prevalent in xeric systems and in mesoxeric systems like the Vermilion River.

On the other hand, the downstream sequence of carbon sources in the Vermilion River still is not consistent with the predicted sequence. While upstream autotrophy was common (but not universal), the development of extensive gallery forests downstream means that terrestrial leaf carbon inputs are not absent but simply displaced longitudinally; that is, the allochthonous to autochthonous sequence from deciduous forests has been inverted but not shortened as the RCC predicts. Downstream declines in photosynthetic rate per unit area coincide with increased inputs of terrestrial litter, and this increase is mirrored in the increasing number of shredders occurring in downstream communities (Wiley and Osborne, unpublished data). Furthermore, the emphasis of the

RCC on the riparian controls of photosynthesis fail to give proper emphasis to the importance of turbidity. As previously discussed, on a strictly statistical basis it can be argued that turbidity was more important than channel shading in explaining the longitudinal distribution of photosynthetic rates.

RCC has difficulty describing rivers like the Vermilion because it relies largely upon generalization rather than mechanism to understand river ecosystem structure. There is simply no single generalization about ecosystem structure that can unambiguously describe the large variety of river systems known to exist. Our goal in theory development should not be to reduce data to a single, all-sufficient, qualitative generalization, but instead it should be to develop a set of quantitative relationships that, when given an array of inputs, can accurately predict the diversity of structures we observe in nature.

In this regard, simple empirical relationships between known, driving variables and system variables of interest may be a far more practical basis for predictive modelling of lotic ecosystem structure. A useful example of the potential power of this approach is the hydraulic geometry commonly used by geomorphologists, hydrologists, and engineers. Stream width, hydraulic depth, mean velocity, and channel cross-sectional area (W, H, V, and C, respectively) can all be expressed as power functions of discharge at a single site (Leopold and Maddock 1953) or for entire watersheds as functions of drainage area and flow exceedence frequency (Stall and Fok 1968, Stall and Yang 1968). For example, basin hydraulic geometry predicts that mean hydraulic depth for any site in a watershed is given by:

$$H = e^{aF} A^{b} \qquad (3)$$

where F is flow frequency, A is drainage area, and a and b are basin-specific coefficients. Similar functions describe related hydraulic variables:

$$W = e^{cF} A^{d}$$
$$V = e^{gF} A^{h}$$
$$C = e^{yF} A^{z}$$

These systems of descriptive equations reflect known relationships between driving variables (*e.g.*, the hydraulic continuity equation: $Q = H \cdot V \cdot D$) and are empirically parameterized for specific watersheds or sites. This approach has provided a set of powerful tools for predicting and describing the hydraulic characteristics of both cross-sections and drainage networks, tools which are frequently used in geomorphological and hydrological research and in fluvial engineering (Thomann 1962).

Simple, empirically constructed models of key biological variables such as Eqs. 1 and 2, when combined with the physical descriptors from hydraulic geometry, may provide a useful prototype of a quantitative theory for stream ecology. For example, Eqs. 1, 2, and 3 can be used to make predictions about the longitudinal profile of community metabolism characteristic of either a deciduous forest or a grassland biome watershed, depending upon the nature of the riparian shading profile (Fig. 3-7). In a deciduous forest profile, we assume shading is highest in the uppermost portions of the watershed, with the canopy becoming more and more open as we move downstream (Fig. 3-7, upper). Resulting GPP/CR ratios are distributed much as the classic RCC described them. Ratios are low in the upper watershed, increase to a maximum in middle-order reaches, and decline slowly as the river becomes larger (Fig. 3-7, lower). In the prairie profile, there is little shading in the headwaters, but, with the development of the gallery forests, light penetration declines in the middle reaches. As the river widens and forces open the canopy above it, shading declines. The metabolic profile derived from this assumption differs substantially from that of the forest. GPP/CR ratios are highest in the headwaters and decline sharply with increasing depth and shading. Loss of shading downstream is offset by increased depth, with only a slight increase in photosynthetic activity. The differences between forested and prairie profiles arise from the different longitudinal profiles of a key, driving variable (in this case, shading or light availability). Note that in this type of model the driving variables need not be distributed in any particular manner. Therefore, this approach does not necessarily posit a "continuum" in either driving variables or lotic system response. Nevertheless, it is consistent with the RCC in that, given a continuous gradient of driving variables (*e.g.*, depth or forest cover), GPP and CR will have predictable longitudinal profiles without strong discontinuities.

We are not suggesting here that any particular equation or model should be adopted, only that models which incorporate key, driving variables can provide a basis for predicting and explaining the diversity of structures, whereas generalized descriptions cannot. Modelling the way in which the pieces (depth, photosynthesis, respiration, light availability, *etc.*) are related to the whole appears to yield greater insight into the structure of stream ecosystems than does modelling the whole itself (as in the RCC's search for a valid generalization or "prototype"). In this case, a moderate dose of reductionism can lead to improved predictability in stream ecosystem theory. Furthermore, for grasslands, savannahs, and possibly xeric sites in general, only analyses at a lower level of organization allow us to see the underlying similarities between these kinds of watersheds and the more familiar mesic watersheds of forested and alpine biomes, because the similarities are mechanistic, not structural. Prairie watersheds and their agricultural descendents are structurally different from forested watersheds, just as desert watersheds are structurally different from alpine watersheds. It is not in spite of, but because of, these differences that the careful comparative study of watersheds will prove valuable in the elucidation of theory for lotic ecology.

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Table 3-1. Land-use/cover distributions for the Salt Fork and Middle Fork basins of the Vermilion River.

Land use	Salt Fork basin Miles^2	Salt Fork basin Percent	Middle Fork basin Miles^2	Middle Fork basin Percent
Agriculture	422	90.8	399	93.0
Urban	26	5.6	5	1.2
Woodland	13	2.8	23	5.4
Wetland	<1	<1.0	1	0.2
Barren	3	<1.0	<1	<0.1
Total	465		428	

Table 3-2. Regression analysis of nephelometric turbidity (log transformed NTU) in the Vermilion River watershed for all stations and all dates. Drainage area is in miles^2; watershed is a dummy variable (0 = Salt Fork, 1 = Middle Fork); flow frequency ranges from 0.0 to 1.0. Model is of the form: $Y = a + b \ln W + c \ln Y + d X$.

Source	DF	SS	MS	F
Regression	3	418.1	139.4	158.2
Residual	427	376.1	0.881	$P < 0.0001$
Total	430	794.166		

$r^2 = 0.53$, standard error = 0.94, coefficient of variation = 41.2

Parameter	Value	T-value	Partial-F
Intercept		2.306	153.44
ln drainage area	0.228	8.38	70.25
ln flow frequency	-4.314	-19.23	369.74
Watershed (dummy)	0.524	5.71	32.64

Table 3- 3. Mean summer (July-September in 1984 and 1985) values for key nutrient, chemical, and biological parameters in the Vermilion River watershed. GPP = gross primary productivity (mg O_2 $m^{-3} \cdot d^{-1}$) and CR = community respiration (mg O_2 $m^{-3} \cdot d^{-1}$). Drainage area is in miles^2. Nitrate (NO_3), soluble reactive phosphorus (SRP), and dissolved oxygen (O_2) concentrations in mg/L. • denotes missing data.

					Diel variation			
Station	Drainage	NO_3	SRP	Turbidity (NTU)	Temp. range	O_2 range	GPP	CR
11	77.0	6.03	4.92	10.29	4.1	2.72	21.71	-47.49
12	83.9	6.40	4.67	4.56	5.0	5.05	20.44	-20.16
13	48.1	4.18	1.92	4.13	9.6	12.44	69.46	-52.40
15	10.0	2.42	0.18	3.96	8.7	6.24	22.28	-38.90
18	22.9	2.70	0.25	5.88	9.9	•	•	•
19	3.7	2.78	0.16	14.36	9.3	4.25	29.66	-29.29
21	90.6	4.08	1.45	2.53	8.2	16.28	100.69	-87.99
22	39.8	3.29	0.18	11.80	•	14.08	84.02	-36.99
24	145.8	4.11	1.04	10.57	6.7	•	•	•
28	11.8	4.55	0.11	13.60	•	6.75	24.93	-72.00
29	8.2	4.10	0.16	22.02	•	•	•	•
30	32.4	4.21	0.11	18.87	•	•	•	•
33	68.0	3.49	0.25	14.12	4.1	•	•	•
34	395.1	4.23	2.34	30.79	2.7	4.42	17.32	-32.28
49	2.7	6.57	0.17	7.96	8.7	2.27	3.92	-18.73
50	85.4	4.86	0.21	13.16	6.8	3.40	38.18	-49.43
53	182.8	4.62	0.99	18.76	3.9	5.59	14.97	-36.80
54	214.3	4.19	0.45	41.72	•	2.09	6.71	-14.04
55	16.3	2.77	0.20	7.22	5.1	•	•	•
57	285.2	3.38	0.52	50.16	•	5.66	25.27	-25.67
59	394.4	3.43	0.44	100.80	6.1	•	•	•
84	239.1	4.99	3.36	9.49	5.2	4.51	2.25	-15.26
86	269.3	4.86	2.93	14.77	4.2	5.01	15.97	-15.69
89	343.8	4.73	2.50	19.07	2.6	.7	2.10	-8.19
90	25.7	6.00	0.32	5.35	•	2.04	7.42	-20.71
91	•	4.29	2.55	22.13	•	•	•	•
92	12.4	4.51	0.15	1.92	•	•	•	•
95	21.4	3.93	0.14	6.05	4.1	•	•	•
96	23.7	4.29	0.15	2.43	6.1	2.41	11.69	-29.95
97	•	3.25	0.51	112.66	4.5	5.48	45.03	-44.30
98	436.8	2.89	0.35	•	3.1	3.65	19.22	-17.75
99	1353.4	3.82	1.23	•	8.2	2.1	9.51	-13.84

Table 3-4. Regression analysis of community photosynthesis (g O_2 $m^{-3} \cdot d^{-1}$) in the Vermilion River watershed for all stations and all dates. Model is of the form: Y+1 = a + b lnW + c lnX + d lnY + e lnZ.

Source	DF	SS	MS	F
Regression	4	229.4	57.4	73.8
Residual	191	148.4	0.778	$P < 0.0001$
Total	195	377.9		

$r^2 = 0.61$, standard error = 0.88, coefficient of variation = 46.3

Parameter	Value	T-value	Partial-F
Intercept	0.804	2.97	
ln turbidity	-0.421	-6.53	42.59
ln depth	-0.683	-5.98	35.79
ln temperature	0.727	9.532	90.85
ln % shaded (+1)	-0.534	-1.578	2.49

Table 3-5. Regression analysis of community photosynthesis (g O_2 $m^{-3} \cdot d^{-1}$) in the Vermilion River watershed. Mean rate from major stations for summer (July-September) only. Model is of the form: Y = a+ b lnW + c lnX + d lnY.

Source	DF	SS	MS	F
Regression	3	17.2	5.73	18.7
Residual	17	5.21	0.307	$P < 0.0001$
Total	20	22.41		

$r^2 = 0.77$, standard error = 0.55, coefficient of variation = 19.3

Parameter	Value	T-value	Partial-F
Intercept	4.008	8.233	
ln turbidity	-.369	-2.70	7.31
ln depth	-.455	-1.84	3.37
ln % shaded (+1)	-2.541	1.837	13.28

Table 3-6. Regression analysis of community respiration rate (g O_2 $m^{-3} \cdot d^{-1}$) in the Vermilion River watershed. Model is of the form: Y+1 = a + b lnW + c lnX + d lnY.

Source	DF	SS	MS	F
Regression	3	73.6	24.5	32.7
Residual	213	159.8	0.8	$P < 0.0001$
Total	216	233.4		

r^2 =0.31, standard error = 0.866, coefficient of variation = 31.7

Parameter	Value	T-value	Partial-F
Intercept	1.608	8.03	
ln depth	-0.098	-0.904	0.817
ln photosynthesis	0.334	5.524	30.518
ln temperature	0.17	1.958	3.834

Table 3-7. Regression analysis of community respiration rate (g O_2 $m^{-3} \cdot d^{-1}$) in the Vermilion River watershed. Mean values for major stations for summer (July-September) only. Model is of the form: Y = a + b lnW + c lnX.

Source	DF	SS	MS	F
Regression	2	6.25	3.12	22.0
Residual	20	2.84	0.14	$\underline{P} < 0.0001$
Total	22	9.07		

r^2 = 0.69, standard error = 0.38, coefficient of variation = 11.5

Parameter	Value	T-value	Partial-F
Intercept	2.034	8.377	
ln depth	-0.276	-1.500	2.25
ln photosynthesis	0.379	3.537	12.508

Table 3-8. Representative values for daily gross photosynthetic rate (GPP) in g O_2 $m^{-2} \cdot d^{-1}$.

GPP	Biome type	Climate	Source
5-18	deciduous forest	mesic	Edwards and Owens (1962)
<1-9	deciduous forest	mesic	Hall (1972)
1-12	deciduous forest	mesic	Bott et al. (1978)
<1-1	coniferous forest	mesic	Naiman and Sedell (1981)
<1-3	coniferous forest	mesic	Duncan and Brusven (1985)
4-48	grassland	mesoxeric	Duffer and Dorris (1966)
1-32	grassland	mesoxeric	Prophet and Ransom (1974)
<1-65	grassland	mesoxeric	present study
8-12	desert	xeric	Minshall et al. (1985)
<1-22	desert	xeric	Busch and Fisher (1981)
14-27	desert	xeric	Cushing and Wolf (1984)

*based on reported hourly values x 10 h of effective daylight.

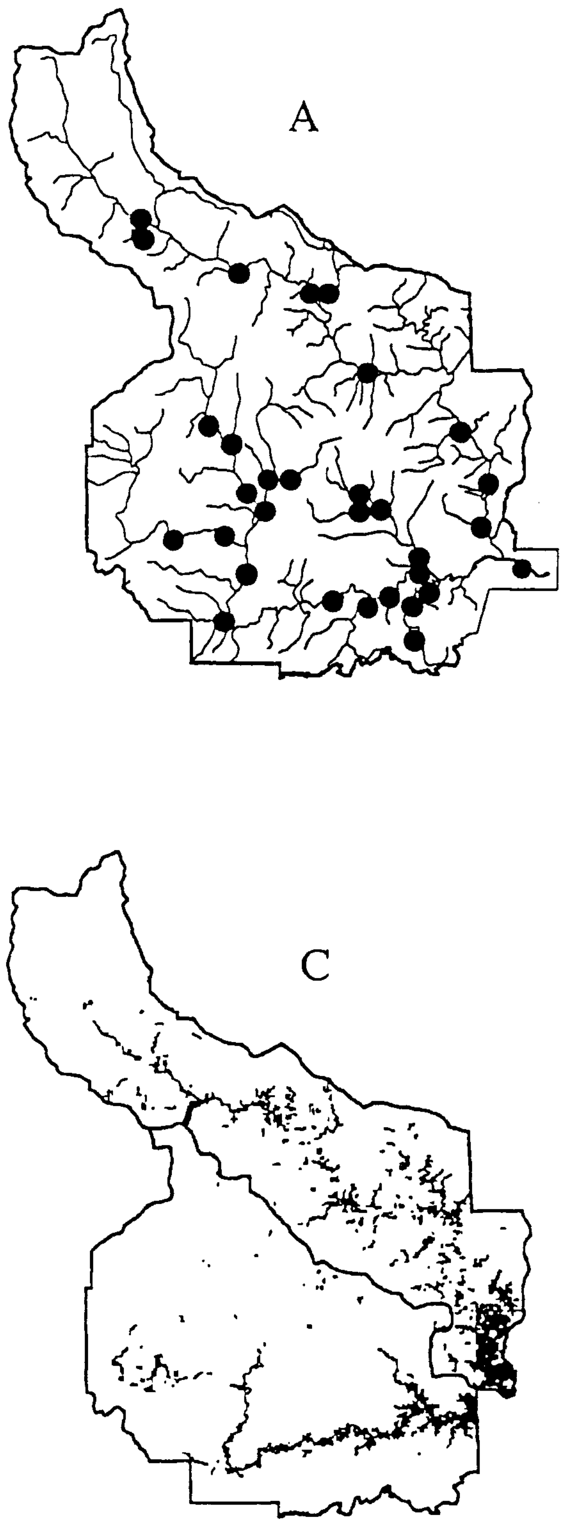

Fig. 3-1. Salt Fork (southwestern) and Middle Fork (northeastern) watersheds. (A) Major sampling locations. (B) Urban and suburban land-use distributions. (C) Forested land distributions. (D) Forest-derived soil associations, based on interpretations following Follmer (1985). A-C are based on photo-interpretation of EROS high altitude photos series 1981 and 1983 (USDA); D is based on data from (Fehrenbacher et al.1984).

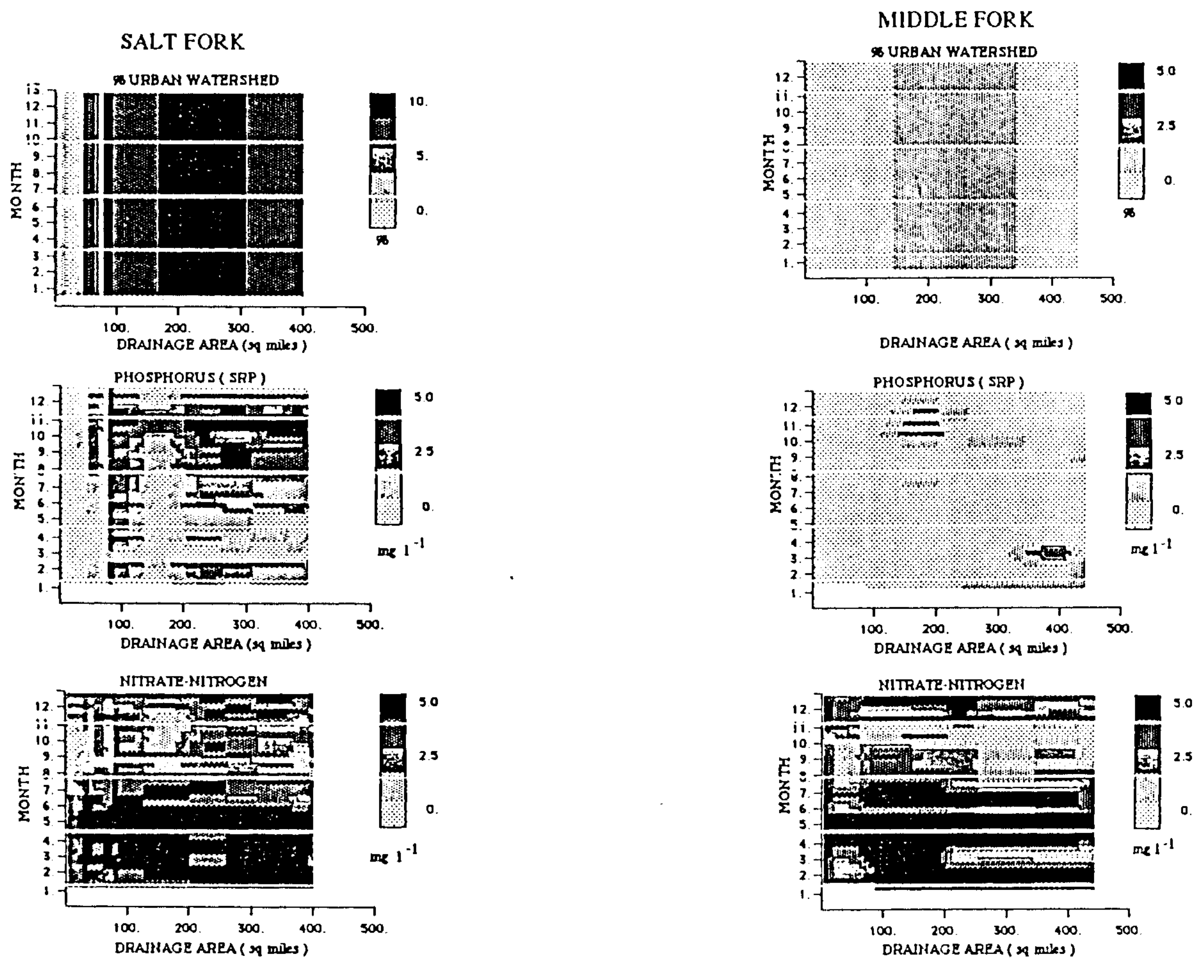

Fig. 3-2. Watershed profiles for urbanization, SRP and nitrate-N concentrations. Digital interpolation is based on 1024 samples collected between January 1984 and December 1985. Vertical axes represents time (month) and horizontal axes represent position in the drainage basin; upstream is to the left and downstream to the right.

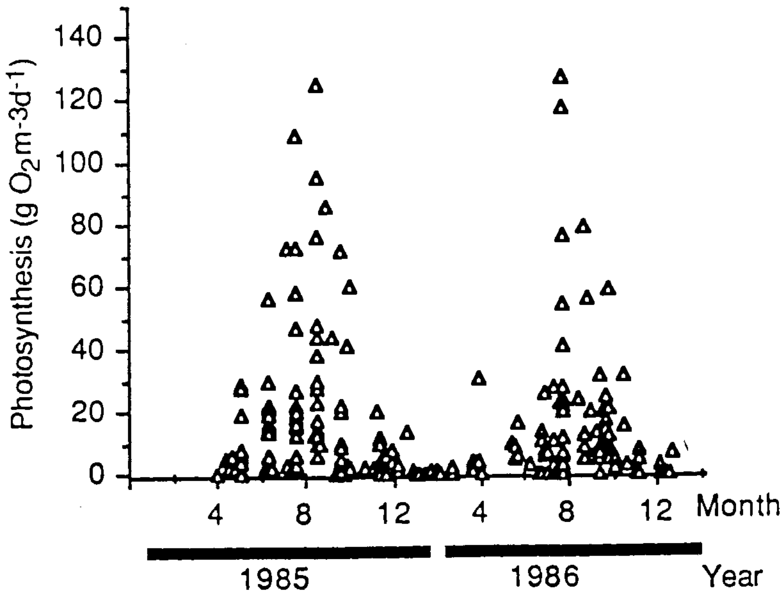

Fig. 3-3. Seasonal distribution of gross primary production rates (g O_2 $m^{-3} \cdot d^{-1}$) for both watersheds and all stations combined.

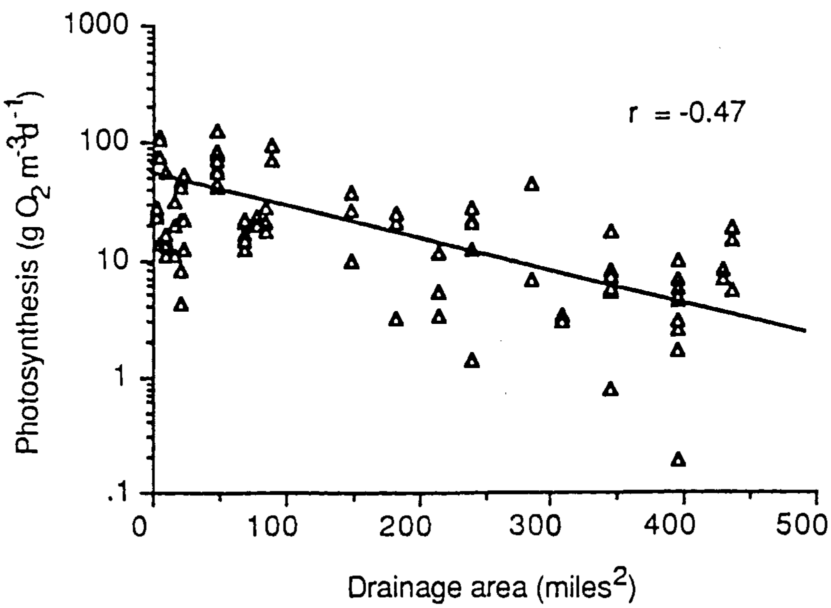

Fig. 3- 4. Longitudinal distribution of gross primary production rates (g O_2 $m^{-3} \cdot d^{-1}$) during summer months (July-September) only.

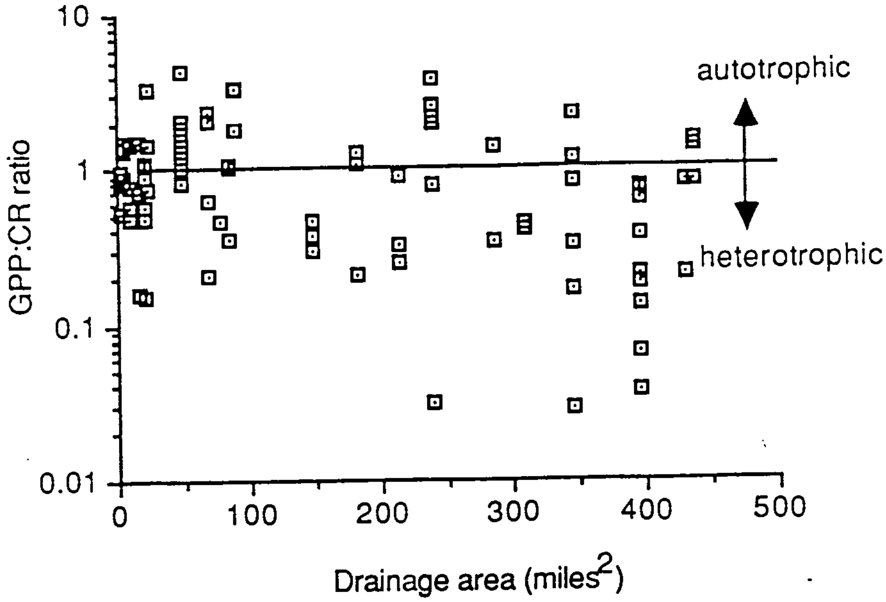

Fig. 3-5. Longitudinal distribution of ratios of total daily gross primary production to total daily community respiration during summer only (July-September).

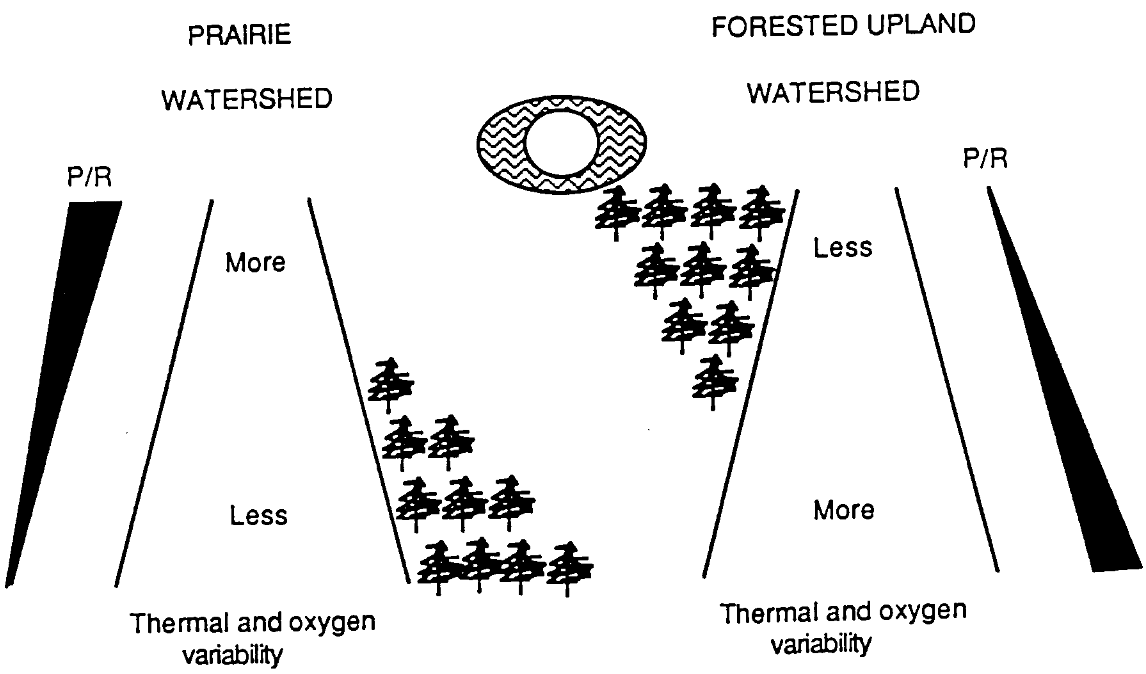

Fig. 3- 6. Summary of organizational differences between grassland and forested watersheds.

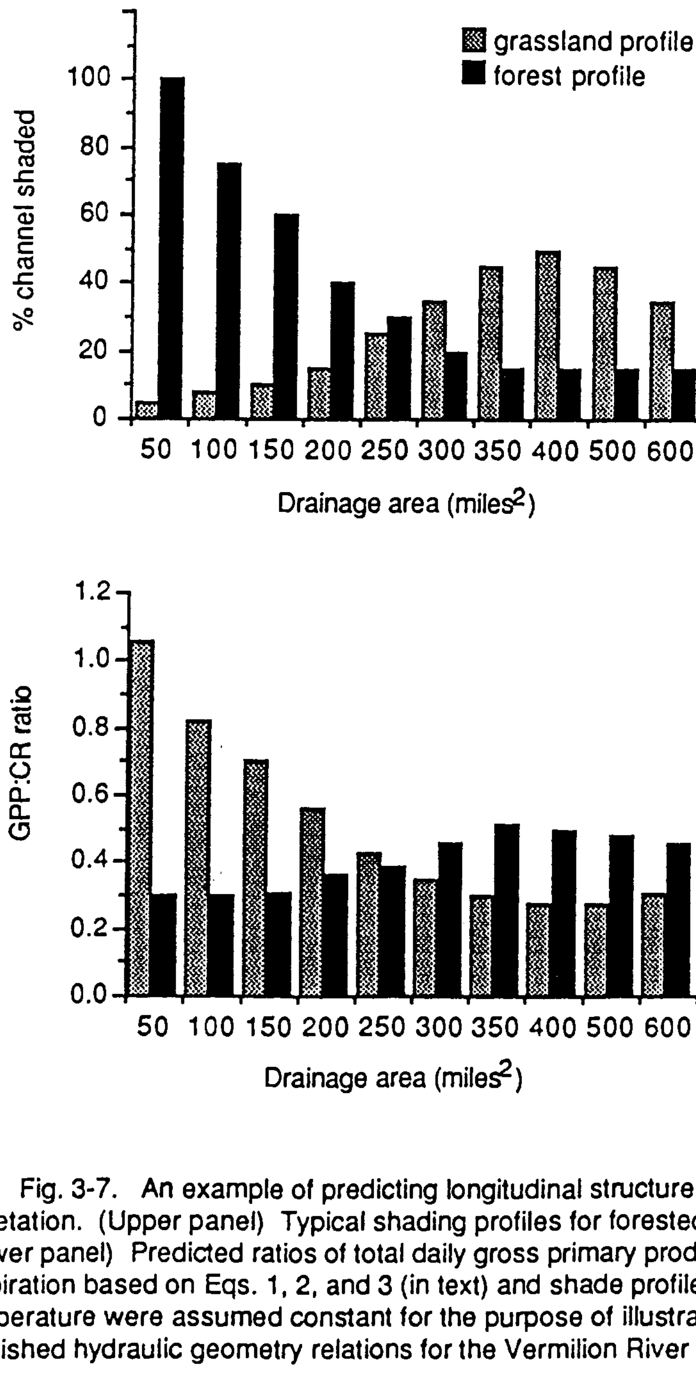

Fig. 3-7. An example of predicting longitudinal structure from profiles of riparian vegetation. (Upper panel) Typical shading profiles for forested and grassland watersheds. (Lower panel) Predicted ratios of total daily gross primary production to total daily community respiration based on Eqs. 1, 2, and 3 (in text) and shade profiles above. Turbidity and temperature were assumed constant for the purpose of illustration, depth was estimated using published hydraulic geometry relations for the Vermilion River (Stall and Fok 1968).